# CARBON NEGATIVE BY 2030
## $CO_2$ removal options for an early corporate buyer

February 2022

## AUTHORS

Briana Mordick Schmidt, Joshuah K. Stolaroff, Sarah E. Baker, Nathan C. Ellebracht, Whitney Kirkendall, Aaron J. Simon, George Peridas, Eric W. Slessarev, Jennifer Pett-Ridge, Simon H. Pang, and Roger D. Aines, Lawrence Livermore National Laboratory, Livermore, CA 94550

Matthew Langholtz, Oak Ridge National Laboratory, Oak Ridge, TN 37830

# ACKNOWLEDGEMENTS

Disclaimer

# TABLE OF CONTENTS

# EXECUTIVE SUMMARY

This report evaluates the options and costs for Microsoft to purchase long-term carbon dioxide ($CO_2$) removal on the approximate scale of 5–6 million metric tons annually by 2030. The scope was largely limited to interventions within the United States, given our resources and expertise. However, some conclusions may be broadly applicable to carbon markets globally, and we note that Microsoft's carbon removal program is not limited to the United States. We examined five pathways for their potential to contribute to Microsoft's carbon removal goal: biomass-based carbon removal and storage (BiCRS and BECCS), soil carbon, forestry, direct air capture (DAC), and carbon mineralization. These five pathways comprise those currently removing $CO_2$ and those we believe are most promising to provide removal at scale in the next decade, within the scope of this analysis.

## Box 1
## CATEGORIZING CARBON REMOVAL

Carbon removal pathways are commonly grouped into two categories: natural and engineered solutions. This is, however, an artificial and even unhelpful distinction. For example, researchers found that climate solutions labeled "natural" are generally received more positively by the public and decision-makers than those labeled "engineered" or "technological," which are often framed as being less desirable, even though on balance the benefits and challenges of each set of solutions may be commensurate [1]. Given this, we have adopted the approach Microsoft has taken to assess their carbon removal based on where it is stored rather than how it is removed; that is, storing it in the biosphere, including plants, trees, and soils, or the geosphere, including rocks underground and minerals [2]. We further suggest a third category of carbon storage in long-lived products; however, this storage pathway was outside the scope of this analysis. This classification system illuminates a key distinguishing feature among carbon removal pathways: durability, i.e., how long removed carbon is likely to stay out of the atmosphere. We do not mean to imply that durability is the only or even most important metric by which carbon removal projects should be judged. Rather, we suggest that, if a categorization system is used at all, it should be informative and reflect a real and meaningful distinction between carbon removal pathways.
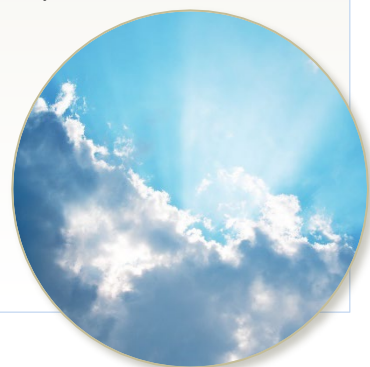
Subsets of these five major pathways were included or excluded based on factors such as data availability, relevance to the stated removal target, and desired characteristics for carbon removal projects. The metrics evaluated differed between carbon removal pathways, as shown in Figure ES-1. For example, "quality" or "acceptability" criteria were key metrics for the biosphere storage pathways but were not explicitly assessed for the geosphere storage pathways. (We note that our analysis was largely conducted prior to the July 2021 publication of the report co-authored by Carbon Direct and Microsoft entitled "Criteria for high-quality carbon removal" (2).) Existing data supported quantitative assessments of removal cost and/or capacity by 2030 for biomass, DAC, carbon mineralization of existing mine tailings, and cover cropping, but such estimates were not supported for most soil and forestry pathways. As a result of these differences and variation in the research questions and analytical methods for each carbon removal type, the key findings for each removal pathway are largely independent of those for the other removal pathways.

## ELEMENTS OF ANALYSIS

| | $ ¥ € £ | Removal Capacity | Durability | Measurability | Additionality |
|---|---|---|---|---|---|
| Biomass Carbon Removal & Storage (BiCRS) | ✓ | ✓ | | | |
| Soils | * | * | ✓ | ✓ | ✓ |
| Forests | | | ✓ | ✓ | ✓ |
| Direct Air Capture (DAC) | ✓ | | | | |
| Carbon Mineralization (CM) | ✓ | ✓ | | | |

*Figure ES-1. The key elements analyzed for each carbon removal pathway are summarized here. As noted above and shown in the figure, the elements varied between pathways. The asterisk indicates that cost and removal capacity were evaluated for cover cropping but not for the other Soils sub-pathways.*

# KEY FINDINGS

**GEOSPHERE STORAGE PATHWAYS**  Cost & Removal Potential by 2030



* Costs shown are for a hypothetical 1 $MtCO_2$/y facility. Total removal capacity in 2030 was not explicitly analyzed, but is essentially unlimited.

*Figure ES-2. Cost and removal potential for the geosphere storage pathways: DAC, BiCRS, and carbon mineralization of existing asbestos mine tailings in the United States and Canada. Total removal capacity in 2030 was not explicitly analyzed for DAC. The cost range given for DAC is for a hypothetical 1 million tons of $CO_2$ per year ($MtCO_2$/y) facility.*

**Of the options assessed in this analysis, BiCRS has the greatest potential to provide durable removal at scale in the next decade, given the low net costs and estimated removal capacity.** We examined two biomass pathways: new-build waste-biomass-gasification-to-hydrogen facilities (*gasification*) and retrofits of existing bioenergy combustion plants with mature carbon capture and storage (CCS) technology (*retrofit*). If limited by current estimates of low-moisture waste biomass in the United States, the gasification pathway could remove more than 200 million tons of $CO_2$ per year ($MtCO_2$/y) for a full-system cost of less than 50 dollars per ton of $CO_2$ ($/$tCO_2$) (including biomass collection, gasification, $CO_2$ transport, and $CO_2$ storage costs). The net cost is low due to the value of hydrogen produced but will require capital investment of about $800M per facility (each with an annual capacity of one million tons of $CO_2$). Retrofits on the most promising existing biomass combustion facilities could remove 12 $MtCO_2$/y for significantly less capital but at a cost of around $100/$tCO_2$.

**Soil carbon and forestry pathways face accounting and durability challenges but can provide important ecosystem, societal, and other benefits.** Soil- and forestry-based carbon removal as currently practiced are relatively high-risk components of a corporate carbon removal portfolio due to challenges related to accounting, durability, additionality, leakage, and other acceptability criteria, as shown in Table ES-1. However, management of soil and forests should still be pursued to achieve other benefits, including reducing greenhouse gas (GHG) emissions from agriculture and protecting existing carbon stocks, sinks, ecosystems, and landscapes, while potentially achieving $CO_2$ removal as a co-benefit. While current soil and forestry removal pathways could theoretically operate at Microsoft's desired scale by 2030 for low to moderate cost, they need stronger standards for accounting and durability to support buyer confidence. In general, verifying the amount of $CO_2$ removal is more difficult for biosphere storage pathways like forests and soils, which involve complex natural systems, and is more straightforward for

geosphere storage pathways. Current practice tends to focus on technical improvements, but robust accounting methods could dramatically improve the market for low-cost, biosphere storage pathways.

**Table ES-1.** Biosphere Storage Pathways – Evaluation of Removal "Quality" or "Acceptability" Metrics.

| Removal Pathway | Sub-pathway | Measurability | Removal vs. avoided emissions | Risk of unaccounted GHG emissions | Additionality | Leakage Risk | Durability |
|---|---|---|---|---|---|---|---|
| Soil | Cover Cropping | ** | ** | ** | *** | ** | ** |
| Soil | Deep-rooted perennial | ** | ** | ** | *** | * | ** |
| Soil | Tillage reduction | * | ** | ** | * | ** | ** |
| Soil | Organic amendments | * | ** | * | * | * | ** |
| Soil | Grazing management | * | ** | * | ** | ** | ** |
| Forestry | Improved management | ** | ** | * | */** | * | */** |
| Forestry | Afforestation | *** | *** | *** | *** | *** | */** |

| Key | | Measurability | Removal vs. avoided emissions | Risk of unaccounted GHG emissions | Additionality | Leakage Risk | Durability |
|---|---|---|---|---|---|---|---|
| | | *low | *avoided only | *high risk | *relatively low | *high risk | *<10 years |
| | | ** intermediate | **mixed | **medium risk | **intermediate | **medium risk | **10-100s years |
| | | ***high | ***removal only | ***low risk | ***relatively high | ***low risk | ***1000s of years |

*Removal quality criteria assessed for biosphere storage pathways: soils and forestry. The metrics evaluated include the following: Measurability: Are established greenhouse gas (GHG) accounting methods available? Are we confident in our ability to measure carbon removal and storage? How well demonstrated are $CO_2$ removals generated by such methods to date? Removal vs. avoided emissions: Does the pathway result in true atmospheric removal, avoided emissions (i.e., preventing $CO_2$ emissions), or a combination of the two? Risk of unaccounted GHG emissions: Does the pathway have sources of GHG emissions that are difficult to quantify or is there uncertainty in identifying all potential sources and sinks? Additionality: Would the removal occur anyway without financial, legal, or other interventions, as measured against a baseline? Leakage risk: Is carbon removal in one place negated by resulting carbon emissions elsewhere, e.g., does the implementation of soil carbon practices on one field lead to more GHG-intensive practices in another field? Durability: How long does the removed $CO_2$ stay out of the atmosphere and can we reliably monitor for reversals?*

**DAC and carbon mineralization are relatively immature and high cost but have large capacity to scale. Investments in pilot projects in the near-term is key to advancing these pathways.** Surprisingly, we anticipate that low-end costs for DAC may not be much higher than those for BiCRS in the coming decade, ranging from $180–450/tCO_2$ for a 1 MtCO_2$/y DAC facility; however, this cost estimate assumes meaningful technology and capacity improvements. Asbestos waste rock in the United States and Canada represents the greatest opportunity for $CO_2$ mineralization of existing mine tailings, with potential to remove 2.75 MtCO_2$/y over 10 years at costs as low as $21–192/tCO_2$; however, land-use requirements would be prohibitive without additional interventions. Projects for these pathways would need to break ground as soon as possible in order to provide removal at scale in the next decade and beyond. Costs for early removal projects are strongly affected by capital availability and development of accounting standards.

While there is no single clear winner in carbon removal technologies, we have identified a combination of removal pathways with the potential to achieve Microsoft's goals by 2030. We find that the supply from these five project types, dominated by the estimated 200 MtCO_2$/y removal from biomass gasification, has the potential to be larger than Microsoft's needs as an individual buyer. This finding is encouraging given the current high demand and short supply in the $CO_2$ removal (CDR) market and demonstrates the availability of durable, moderate-cost removals for other buyers in the coming decade, although it still depends on considerable market development efforts.

The potential for new carbon removal technologies to be impactful in the next decade is limited by technological maturity, deployment rate, and accounting methods. To make a significant contribution by 2030, a technology will have to reach development, demonstration, and market growth before that time, as shown in shown in Figure ES-3 and must have an accepted accounting method reflecting established quality or acceptability criteria.
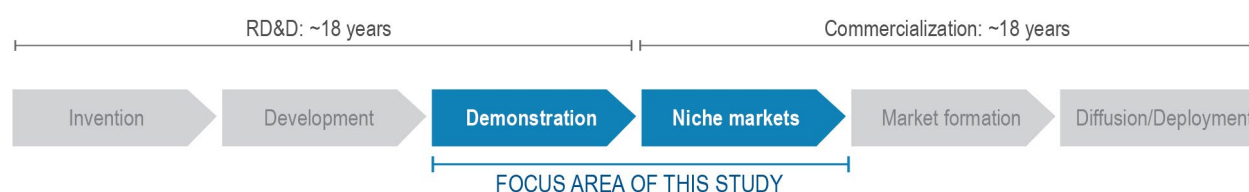


**Figure ES-3.** *Timescale for energy technology innovation from invention to commercialization. Derived from (3).*

The development of a carbon removal market is an unprecedented race against the clock, as "the median time taken from invention to widespread commercialization [of other technologies has typically been] 32 years" (with a range of 20–80 years) (*3*). **In light of this timeline, we expect that the carbon removal market in 2030 will very likely comprise technologies that have already been piloted today and perhaps some that will be piloted by 2025.** Looking ahead, strategic investments are needed to help catalyze deployment of technologically immature removal pathways and also to incentivize development of stringent monitoring and accounting practices for biosphere storage pathways. It is important to note that negative emissions cannot be a substitute for avoiding and reducing emissions if

we are to meet the goal of limiting global warming to less than 1.5–2 °C. Rather, carbon removal technologies play a critical complementary role to replacing polluting sources by offsetting emissions from sources that are truly difficult to decarbonize and by removing historic emissions to prevent future warming.

## DETAILED FINDINGS

### Biomass Carbon Removal and Storage (BiCRS)

We assessed promising near-term (2030) opportunities for biomass carbon removal and storage (BiCRS) (*4*) deployment in the United States, focusing on two technologically mature pathways that are potentially deployable at scale by 2030: 1) waste biomass gasification to produce liquid hydrogen and $CO_2$ for sequestration and 2) retrofit of existing biomass-combustion facilities with carbon capture. We chose to focus on the gasification pathway because we previously identified gasification to hydrogen as one of the highest carbon yielding, lowest cost per ton of $CO_2$ thermochemical conversion routes (*5*). However, new construction of a large facility carries risks, including securing stable sources of biomass, biomass transport logistics, and securing permits and capital for construction. Therefore, for comparison with an alternative near-term BiCRS technology, we analyzed costs and other considerations involved with the lower risk pathway of retrofitting existing biomass-combustion facilities.

The gasification pathway requires siting and constructing a new facility. Based on previous analysis, we assumed that gasification facilities would be located in regions with favorable geology for $CO_2$ sequestration, with the goal of minimizing $CO_2$ transport costs from the gasification facility to the $CO_2$ injection sites. We first sought to identify the most promising U.S. geologic storage regions, as well as U.S. regions with biomass types, densities, and locations suitable for providing biomass feedstocks for gasification. In contrast, the retrofit pathway assumes the biomass supply and agreements are in place; for this pathway, we evaluated existing facilities on the basis of scale, type of biomass (biogenic fraction of carbon in fuel streams), and effects of adding carbon capture to the net power generation. We compared these two pathways on the basis of available $CO_2$ removal capacity, capital cost per plant, and cost per ton of $CO_2$.

The figure and caption depict our key findings for carbon removal costs for the gasification and retrofit pathways. We also analyzed the carbon-removal capacity for each pathway.
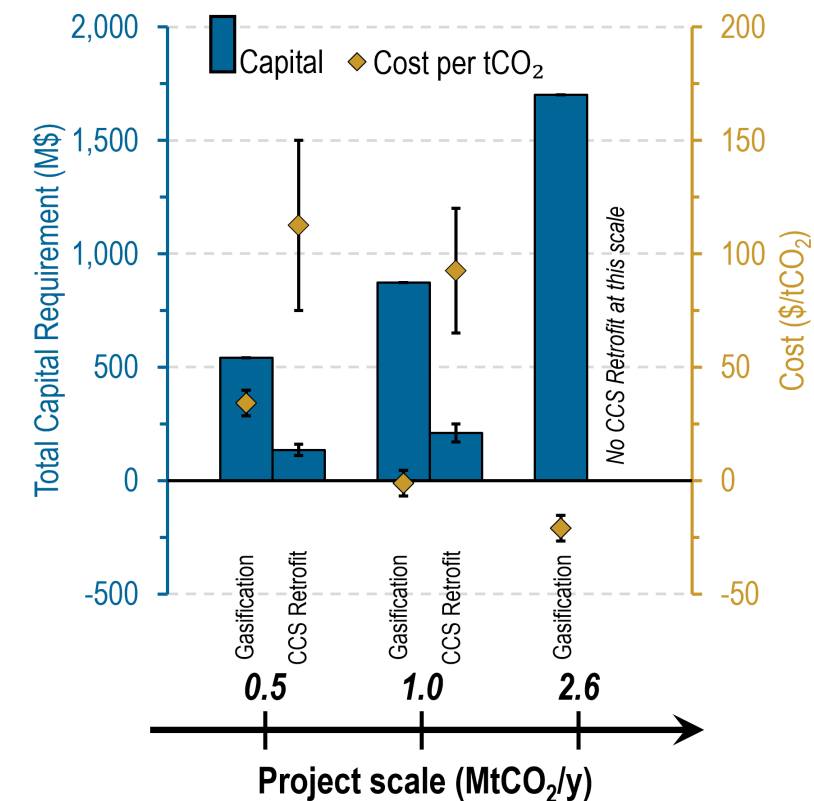
**Figure ES-4.** *Capital costs and $CO_2$ removal costs ($/tCO_2$) for the two BiCRS pathways analyzed in this chapter: 1) the new construction of biomass gasification plants with sale of liquid hydrogen and 2) the retrofit of existing biomass-combustion facilities with carbon capture. Our results indicate that the cost per ton (indicated by markers) of $CO_2$ for retrofits is higher (2–4 fold) than for new-build gasification plants (around $100/tCO_2$ vs $<$50/tCO_2$). However, initial capital investment (indicated by bars) for retrofits is significantly lower than for construction of new gasification plants.*

**Overall, we found that existing U.S. waste biomass could provide at least 200 million tons of negative-emissions $CO_2$ via the gasification pathway. Build-out of the retrofit pathway to the most-promising (<$200/tCO_2$) existing facilities would provide 12 million tons of negative-emissions $CO_2$ per year for significantly less capital expenditure than the gasification pathway.** However, our analysis indicated that the *cost per ton of $CO_2$ for the retrofit pathway would be significantly higher than for the gasification pathway.**

The most-promising locations for new-build gasification plants, corresponding to high waste-biomass densities and low transport costs, are in Central California, the Gulf Coast, Florida, and some areas in the Midwest and Northeast. Biomass-combustion facility retrofit areas of interest in the United States include the Southeast (wood, agriculture, and municipal solid waste (MSW)), the Northeast (MSW), and California (wood, MSW, and agriculture).

## Soils

Agricultural soils have lost a vast amount of carbon to the atmosphere over human history (*6*). Even partially reversing this loss would remove a significant quantity of carbon from the atmosphere, thus

slowing climate change (*7–9*). On the other hand, soils are dynamic natural systems that respond differently to management in different environments; hence efforts to increase soil carbon storage do not reliably yield net removal of $CO_2$ from the atmosphere in every agricultural or geographic context. Our analysis considers how these complexities influence the potential for soils to sequester carbon.

In addition to the broad criteria for carbon removal outlined above, we propose specific criteria that define effective soil-based carbon removal. For our analysis, we applied these criteria to five soil-management practices that have putative potential to achieve carbon removal: cover cropping, conversion of annual cropland to perennials, tillage reduction, organic amendments, and improved grazing management. We then mapped potential low-cost opportunities for soil-based climate mitigation at a county-level across the United States using the COMET Planner tool (*10*). Finally, we considered unresolved challenges related to soil carbon in the context of carbon credits.

***Our major finding is that establishing verifiable and permanent $CO_2$ removal is challenging across all soil-based strategies; consequently soil-based carbon removal is a relatively high-risk method for offsetting $CO_2$ emissions. However, this risk does not obviate the need to manage soil in order to reduce GHG emissions from agriculture, while potentially achieving $CO_2$ removal as a co-benefit.***

We developed first-order estimates for capacity and future costs of soil-based climate mitigation potential (defined as *removal plus avoided emissions*). We focused on cover cropping, which we ranked as having a more clearly demonstrated climate mitigation potential than other practices, based on the soil-specific quality criteria. Within the South-Central United States, cropping might facilitate avoided emissions or $CO_2$ removal equivalent to 6 million tons of $CO_2$-equivalent per year over a 10-year period (MtCO2e/y; $CO_2$-equivalent is a measure of $CO_2$ and other GHGs, where other GHGs are converted to an equivalent amount of $CO_2$ using the Global Warming Potential of each gas). We estimate that costs of avoided emissions and removal associated with cover cropping in the South-Central United States are between \$30 and 100/tCO2e (9% discount rate) or \$200/tCO2e (1% discount rate) assuming a 100-year target for permanence. Costs are sensitive to the risk of reversion to conventional management, which is not well constrained. ***We stress that these estimates combine avoided emissions and removal, which are not clearly differentiated in current soil carbon crediting protocols (6) and which should not be treated as equivalent quantities (7).*** The primary effect of best-practice management of agricultural soils is often reduced emissions, not net removal of carbon from the atmosphere (8, 9).

## Forestry

Forestry is an often-cited intervention for avoiding carbon emissions and/or removing carbon from the atmosphere. Aside from the ecological benefits of preserving or restoring forests, forestry offsets appear attractive from a price standpoint, as they typically cost less than technological carbon removal solutions. Among natural climate solutions, forestry occupies central stage because of its large potential scale. Here, we investigated the physical feasibility of removing $CO_2$ from the atmosphere at a multi-million-ton-per-year scale through forestry, i.e., a scale that is meaningful for corporate first movers in the carbon removal space. We also looked at related challenges, such as transactional volume, baseline definition, additionality, leakage, and durability. This analysis was based on literature review and consultation with sectoral and topic experts.

Managing forests for the purpose of carbon removal can have ecological and landscape co-benefits, making the pursuit of projects desirable for reasons beyond carbon. However, despite the large theoretical potential to sequester $CO_2$ in U.S. forests, there are several challenges that stand in the way of achieving removals at the multi-million-ton-per-year scale through forestry. These include the potentially very large number of landowners and transactions involved—a logistical challenge that may be mitigated in the future through technology. They also include difficulties in defining project baselines, establishing additionality, and quantifying credits appropriately, concerns over induced impacts elsewhere (leakage), and a relatively high risk of reversal—accounting challenges that will require significant effort from project practitioners, researchers, registries, and governments to overcome. Technology and improved project governance and accounting structures may be able to assist with some of these challenges but likely not with durability concerns.

***Therefore, despite its apparent attractiveness, the forestry sector must address challenges related to definition of baselines, additionality, leakage, and durability (including risk of reversal due to fire) in order to be a sound backbone for corporate carbon removals at the needed scale. As such, forest carbon credits should be used judiciously, with individualized due diligence at the project level, and under more thorough safeguards and frameworks than are publicly available today.***

***As with soil carbon sequestration, the challenges of forest carbon projects do not obviate the need to protect and manage forests to maintain existing carbon stocks, sinks, ecosystems, and landscapes, while potentially achieving other additional co-benefits.***

## Direct Air Capture (DAC)

DAC is a straightforward method of providing negative emissions that does not suffer from questions of additionality or durability. Compared to other technology options, DAC is relatively nascent with limited global deployment and, as such, has a high cost. We evaluated several classes of DAC technologies in terms of their energy requirements and cost to produce $CO_2$ suitable for sequestration. All pathways have a significant energy requirement many times the thermodynamic minimum, likely leading to concurrent buildout of dedicated energy sources, particularly in the long-term when the total quantity of deployed DAC capacity is large.

Areas of opportunity in research and development for all classes of DAC technology are available that will help to reduce either the cost or the uncertainty in the cost. Broadly, these research areas are related to improving the lifetime and $CO_2$ throughput of the materials, increasing the energy efficiency of processes, and reducing the cost of the raw chemicals required for consumables. In addition, due to the low total capacity of DAC currently deployed, all pathways will benefit from expanded deployment and will see cost reductions via learning-by-doing.

***We estimate that costs in 2030 will range from approximately \$180–450/t$CO_2$ for a 1 Mt$CO_2$/y DAC facility. This cost estimate assumes improvements in sorbent lifetime and roughly 8 Mt$CO_2$/y deployed capacity for each individual DAC pathway, indicating that projects will need to begin breaking ground as soon as possible, with significant buildout over the next several years compared to the currently deployed capacity. At the Mt-scale, DAC facilities operating at current energy efficiencies will require a***

***250-MW dedicated energy source, suggesting that local energy infrastructure is a serious consideration for locating DAC facilities.***

## Carbon Mineralization

Carbon mineralization seeks to replicate and accelerate Earth's natural process of chemical weathering, which transfers $CO_2$ from the atmosphere into rocks. The removal potential of carbon mineralization is essentially unlimited given the vast quantities of suitable rocks available. While our fundamental understanding of the processes is relatively mature, practical application is lacking and field projects are a critical next step to scaling up.

This analysis considers three carbon mineralization pathways, each of which is intended to achieve negative emissions: 1) surficial carbon mineralization of existing fine-grained mine wastes (e.g., "tailings") at inactive mines only, due to concerns about the environmental and social impacts of mining and to avoid the need to offset ongoing emissions from mining activities, 2) *in situ* mineralization as geologic carbon storage, in which another CDR technology, such as DAC or BiCRS, removes $CO_2$ from the atmosphere, and 3) *in situ* mineralization as standalone CDR, in which the mineralization process itself removes $CO_2$ from the atmosphere.

We developed an illustrative estimate of the cost and removal potential associated with existing mine tailings in the United States and Canada. We also qualitatively assessed the prospects for *in situ* mineralization to provide removal at scale before 2030.

***Asbestos tailings in the United States and Canada represent the greatest opportunity for mineralization of existing mine tailings and may have the ability to remove up to ~750 MtCO₂.*** Over 10 years, the most reactive fraction of minerals in these tailings in Canada, Vermont, and California could remove ~27, 0.4, and 0.34 $MtCO_2$ for costs of ~$75, 75, and 21/$tCO_2$, respectively. However, the land-use requirements to achieve this level of removal could be prohibitive—a 1-cm-thick layer of the 2 Gt of tailings in Southern Quebec would cover 76,900 km$^2$ or about 40% of the area of Washington State. ***Additional interventions, such as improved grinding, mixing, and spreading, and better approaches in general than simply spreading tailings over bare ground would be needed to reduce land use.*** On the order of 100–400 million tons of tailings would be required to remove Microsoft's 2030 goal of 5–6 $MtCO_2$ when relying on only the most reactive fraction of minerals, which make up ~10 wt% of suitable mine tailings at most and typically make up much less. Additional interventions such as those listed above would be needed to increase the reaction rate of the slower reacting but more abundant minerals.

*In situ* mineralization as storage for other CDR technologies is the most mature form of carbon mineralization and can expand the geographic range of these technologies to places where traditional saline geologic storage is not available. Reported costs of mineralization storage (~$17–24/$tCO_2$) still exceeds the average reported cost of traditional storage in sedimentary formations (~$7–13/$tCO_2$) but should be weighed against unmonetized benefits of mineralization storage, such as the potential for greatly reduced monitoring requirements due to the high security of $CO_2$ storage once mineralization is

complete. However, it is important to note this form of mineralization does not itself remove $CO_2$ from the atmosphere; it is simply another form of geologic storage.

*In situ* mineralization as a standalone CDR technology has enormous potential to scale but is immature and unlikely to provide removal at scale prior to 2030.

# CHAPTER 1. INTRODUCTION

## BACKGROUND AND GOALS

By 2020, over a third of Fortune 500 companies had adopted ambitious greenhouse gas (GHG) emissions reductions targets (*11*). Over 1000 companies globally have partnered with the Science Based Targets Initiative to make science-aligned climate commitments (*12*). Among the more aggressive targets, more than 100 companies have signed the Climate Pledge, committing to achieve net-zero emissions by 2040. Microsoft Corporation is among those companies, committing to be carbon-negative by 2030 (*13*) by reducing its value chain emissions by more than half and removing the rest and committing to removing its historical emissions by 2050.

At the global scale, we need to turn emissions net-negative by roughly 2050 to meet the goal of limiting global warming to less than 1.5 °C (*14–16*). To achieve this, in turn, hundreds of gigatons of carbon removal are required by 2050 in most scenarios to offset residual emissions in hard-to-mitigate sectors and, eventually, to remove past $CO_2$ emissions from the atmosphere. Similarly, for an organization to achieve net-zero, they will generally have to employ some amount of carbon removal for the foreseeable future. Microsoft estimates they will need roughly 5–6 million tons of carbon removal per year in 2030 to achieve net-zero emissions and more to remove their historical emissions by 2050.

Microsoft asked Lawrence Livermore National Laboratory (LLNL) to assess the role that $CO_2$ removal (CDR) could play in meeting its 2030 climate goals, including identifying key technology pathways that can provide removal at the desired scale by 2030. The quantity of negative emissions required by Microsoft is small compared to estimates of the eventual global market for carbon removal and compared to the technical and physical limits of prominent carbon removal technologies (*17*). However, Microsoft's expected demand far exceeds the current global supply. The short time horizon to 2030 invites a closer look at the options.

Markets for carbon offsets have been running since the 1990s and although some of the interventions might be considered carbon removal in retrospect, the markets for carbon removal *per se* are brand new. This presents several related challenges: 1) growth of the market to meet the demand by Microsoft and other organizations with similar needs, 2) the quality or acceptability of the carbon removal, since few methods are proven over time, and 3) the cost of carbon removal compared with perceived benefits or other mitigation options.

In this report, we assess the opportunities for carbon removal by the year 2030. We consider market-based mechanisms that satisfy the stated criteria of Microsoft and other organizations seeking high-quality carbon removal to achieve net-zero or net-negative emissions goals (we note that our analysis was largely conducted prior to the July 2021 publication of the report co-authored by Carbon Direct and Microsoft entitled "Criteria for high-quality carbon removal" (*2*)). Our scope is mostly limited to interventions within the United States, though we discuss international opportunities for some technologies. We estimate the costs and quantities available for a set of the most promising

interventions and discuss the research and investment strategies that can help build a more robust set of options by 2030.

## FOCUS AREAS OF THIS REPORT

The technologies we considered for this report are those that can generate negative carbon emissions—in other words, they can remove $CO_2$ from the atmosphere. These are distinct from, although they may overlap with, technologies that replace polluting sources (e.g., replacing a natural gas-fired power plant with solar energy) or capture $CO_2$ before it reaches the atmosphere (e.g., installing $CO_2$ capture equipment on a natural gas-fired power plant). These latter technologies avoid emissions, as opposed to generating negative emissions.

It is important to note that negative emissions cannot be a substitute for avoiding and reducing emissions if we are to meet the goal of limiting global warming to less than 1.5–2 °C. Rather, carbon removal technologies play a critical complementary role to replacing polluting sources by:

1) Offsetting emissions from sources that are truly difficult to decarbonize, either because alternatives do not exist at all or at scale or because eliminating them would cause unacceptable impacts, like food shortages
2) Removing historical emissions to prevent future warming from $CO_2$ already in the atmosphere

In the case of Microsoft and other companies committed to reaching net-zero, carbon removal technologies are also key to addressing emissions caused by activities outside of the company's direct control, known as "Scope 3" emissions. These include emissions caused by its suppliers or customers and typically constitute a company's largest source of emissions. Importantly, Microsoft has committed to *reduce* all its value chain emissions (Scopes 1, 2, and 3) by more than half by 2030 and to *remove* $CO_2$ in an amount equal to only its residual, hard-to-abate emissions. In recognition of the infancy of the carbon removal market today, Microsoft has already begun to purchase and invest in carbon removal solutions to help jump-start supply.

Five classes of carbon removal are considered for this report: biomass-based carbon removal and storage (BiCRS and BECCS), soil carbon, forestry, direct air capture (DAC), and carbon mineralization. These five classes are a combination of those currently removing $CO_2$ and those we believe are most promising for providing removal at scale in the next decade. As is discussed in detail in subsequent chapters, subsets of these five major classes were included or excluded based on factors including data availability, relevance to the stated removal target, and the desired characteristics for carbon removal projects. Some categories of carbon removal are not assessed here—for example wetlands restoration, biochar, bio-oil injection, and ocean alkalinization—because not enough information was available on them or because they could not be adequately evaluated within the scope of the report.

The metrics evaluated differed between carbon removal pathways. For example, "quality" or "acceptability" criteria were key metrics for the forestry and soil carbon pathways but were not explicitly assessed for the other three. Existing data supported quantitative assessments of removal cost and capacity by 2030 for biomass, DAC, carbon mineralization of existing mine tailings, and cover

cropping, but such estimates were not supported for most soil and forestry pathways. The key research questions and analytical methods varied for each carbon removal type, which is reflected in the fact that the content and conclusions of each chapter are largely independent of the other chapters.

We have for the most part focused on the opportunities and challenges of each removal pathway individually. However, in practice, each pathway will require a coordinated system of technologies and practices to provide removal at scale. For example, DAC must be paired with technologies to either utilize or store the removed $CO_2$ and must be supported by a robust supply of renewable energy. These broader-scale system issues are largely beyond the scope of this analysis.

## POTENTIAL FOR NEW TECHNOLOGIES

The carbon removal market exists today, which is substantially a result of purchases by Microsoft and a handful of other early corporate purchasers. However, the current scale and quality of removal is not adequate to meet the growing demand of these organizations, let alone many new buyers to come. In this report, we assess whether new technologies can meet the demand gap by 2030, with a focus on where investment can accelerate promising new technologies.

The potential for new carbon removal technologies to contribute to reducing atmospheric $CO_2$ in the next decade is limited by both technological maturity and deployment rate. To make a significant contribution to planetary carbon removal needs by 2030, a technology will have to reach development, demonstration, and market growth by that time, as shown in Figure 1-1.
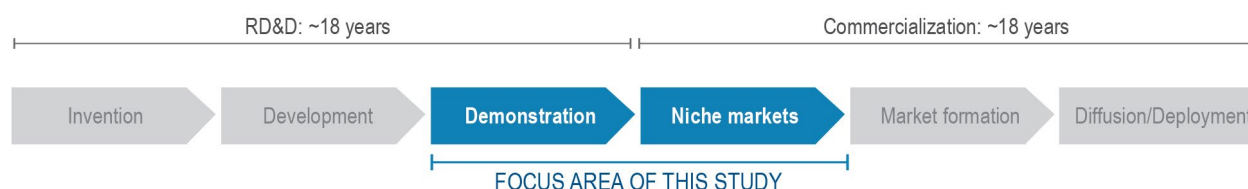


**Figure 1-1.** *Timeline for new technologies to progress from invention to deployment. Derived from (3). Our analysis is largely focused on technologies that have already been piloted, such that they can achieve market growth by 2030.*

As noted by Gross et al. 2018 (*3*) "the median time taken from invention to widespread commercialization [of other technologies has typically been] 32 years" (with a range of 20–80 years). Successful carbon removal market development is an unprecedented race against the clock.

**In light of this timeline, we expect that the carbon removal market in 2030 will very likely comprise technologies that have already been piloted today and perhaps some that will be piloted by 2025.** In the following chapters, we focus on the technologies that are close to pilot demonstrations or even further along their developmental paths. For investment activities to change the 2030 landscape, they should focus on pilot projects in the next few years and on market building for technologies in the later stages of development.

# CHAPTER 2. BIOMASS CARBON REMOVAL AND STORAGE

## SUMMARY

We assessed promising near-term (2030) opportunities for biomass carbon removal and storage (BiCRS) (*4*) deployment in the United States, focusing on two technologically mature pathways that are potentially deployable at scale by 2030: 1) waste biomass gasification to produce liquid hydrogen and $CO_2$ for sequestration and 2) retrofit of existing biomass-combustion facilities with carbon capture. We chose to focus on the gasification pathway because we previously identified gasification to hydrogen as one of the highest carbon yielding, lowest cost per ton of $CO_2$ thermochemical conversion routes (*5*). However, new construction of a large facility carries risks, including securing stable sources of biomass, biomass transport logistics, and securing permits and capital for construction. Therefore, for comparison with an alternative near-term BiCRS technology, we analyzed costs and other considerations involved with the lower risk pathway of retrofitting existing biomass-combustion facilities.

The gasification pathway requires siting and constructing a new facility. Based on previous analysis, we assumed that gasification facilities would be located in regions with favorable geology for $CO_2$ sequestration, with the goal of minimizing $CO_2$ transport costs from the gasification facility to $CO_2$ injection sites. We first sought to identify the most promising U.S. geologic storage regions, as well as U.S. regions with biomass types, densities, and locations suitable for providing biomass feedstocks for gasification. In contrast, the retrofit pathway assumes that the biomass supply and agreements are in place; for the retrofit pathway, we evaluated existing facilities on the basis of scale, type of biomass (biogenic fraction of carbon in fuel streams), and how adding carbon capture affects net power generation. We compared these two pathways on the basis of available $CO_2$ removal capacity, capital cost per plant, and cost per ton of $CO_2$.
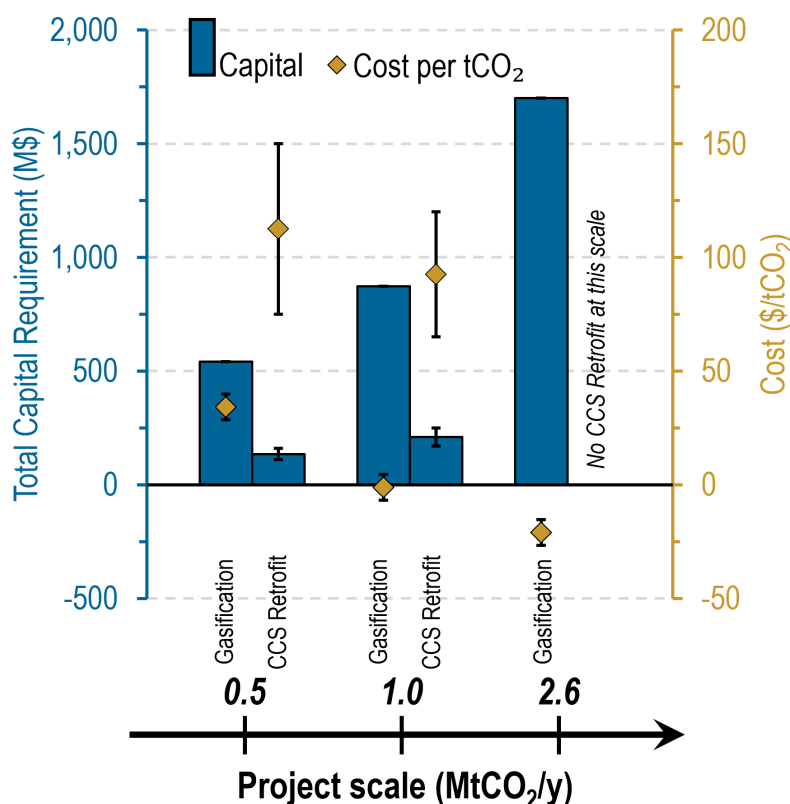
**Figure 2-1  BiCRS Key Finding.** *Capital costs and $CO_2$ removal costs ($/tCO_2$) for the two BiCRS pathways analyzed in this chapter: 1) new construction of biomass gasification plants with sale of liquid hydrogen and 2) retrofit of existing biomass-combustion facilities with carbon capture. Our results indicate that the cost per ton (indicated by markers) of $CO_2$ for retrofits is higher (2–4 fold) than for new-build gasification plants (around $100/tCO_2$ vs <$50/tCO_2$). However, initial capital investment (indicated by bars) for retrofits is significantly lower than for construction of new gasification plants.*

The figure and caption depict our key findings for carbon removal costs for the gasification and retrofit pathways. We also analyzed the carbon-removal capacity for each pathway.

*Overall, we found that existing U.S. waste biomass could provide at least 200 million tons of negative-emissions $CO_2$ via the gasification pathway. Build-out of the retrofit pathway to the most-promising (<$200/tCO_2$) existing facilities would provide 12 million tons of negative-emissions $CO_2$ per year for significantly less capital expenditure than the gasification pathway.* However, our analysis indicated that the *cost per ton of $CO_2$ for the retrofit pathway would be significantly higher than for the gasification pathway.*

The most-promising locations for new-build gasification plants, corresponding to high waste-biomass densities and low transport costs, are in Central California, the Gulf Coast, Florida, and some areas in the Midwest and Northeast. Biomass-combustion facility retrofit areas of interest in the United States include the Southeast (wood, agriculture, and municipal solid waste (MSW)), the Northeast (MSW), and California (wood, MSW, and agriculture).

# INTRODUCTION

Bioenergy with carbon capture and storage (BECCS) is a prominent pathway to negative emissions, with inclusion in major reports (including IPCC 2018 (*18*)) in which the necessary negative emissions to achieve <1.5 °C warming are achieved in the integrated assessment models through a combination of BECCS, afforestation, and direct air capture (DAC) (*19*). BECCS, as conventionally envisioned, uses biomass combustion to generate electricity integrated with capture and geologic storage of the resulting $CO_2$. If the amount of carbon stored underground exceeds the $CO_2$ equivalents emitted during biomass cultivation, transport, conversion, and storage, then the pathway is considered to have removed net $CO_2$ from the atmosphere.
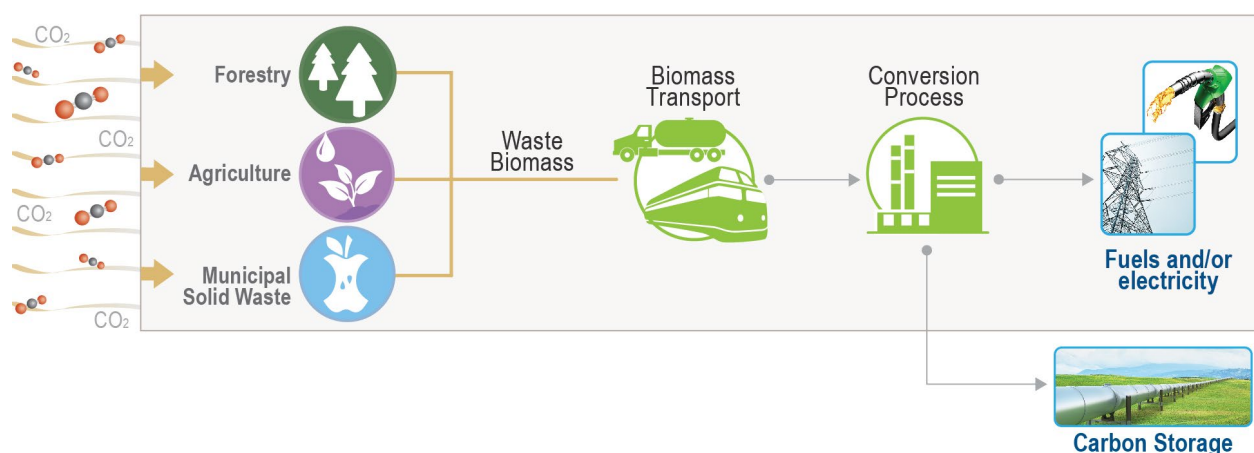


**Figure 2-2.** *Schematic depicting biomass carbon removal and (optional) production of fuels or electricity in BiCRS negative-emissions pathways.*

Despite extensive system-level studies on benefits and sustainability considerations around BECCS, large-scale deployment of BECCS based upon biomass combustion has been limited. As of 2019, five facilities around the world were capturing $CO_2$ resulting from processes for biomass-produced energy and/or fuels, with the majority of the 1.5 $MtCO_2$/y being captured at a fermentation plant—the Archer Daniels Midland (ADM) corn ethanol production facility in Illinois—rather than at a combustion plant (*20*). To date (as of March 2021), the ADM facility has stored 3.4 $MtCO_2$ in on-site geologic wells (*21*). Further, the United States is *losing* capacity in biomass combustion facilities due to air quality concerns from older combustion facilities and the poor economics of electricity generation from biomass relative to other renewables. However, the closing of biomass combustion facilities can force biomass producers to dispose of their waste biomass using pile burns, which worsen air quality and waste the energy and carbon in the resource. Some researchers have concerns around BECCS, focused on the impacts of production and collection of biomass feedstocks at the required gigaton scales that are assumed in Integrated Assessment Models for global decarbonization. These concerns include negative impacts around irrigation, land and fertilizer use, maintaining ecosystems and biodiversity, and competition with food production (*22, 23*). In contrast, others contend that converting some marginal cropland from annual crops to perennial biomass crops could improve soil conservation, water quality, wildlife habitat, and rural economies with minimal competition with food production (*24–27*).

The use of waste biomass rather than purpose-grown biomass is the focus of this report. Since we only consider waste biomass outside of current uses, including current requirements for soil health and reduced erosion, the risk of BECCS causing negative environmental impacts is significantly reduced. However, an "implementation gap" does remain: despite the fact that BECCS processes are technologically mature, BECCS will remain only the subject of systems-level study and debate unless this implementation gap is overcome with 1) immediate action, 2) successful small-scale demonstrations, 3) implementation of necessary incentives, and 4) favorable social acceptance and awareness of societal co-benefits (23). The logistics around large-scale transport of biomass to a conversion facility, followed by capture of process carbon and transport and underground storage of captured $CO_2$, presents additional challenges to large-scale deployment of BECCS.

**Recently, we and others (4) have introduced a more general term for the use of biomass to achieve negative emissions: biomass carbon removal and storage (BiCRS). This term emphasizes *the value of biomass for CARBON REMOVAL rather than for energy*** and, thus, includes the production of biomass carbon suitable for sequestration alone (e.g., biochar), as well as the production of carbon-negative liquid and gaseous fuels. The benefits of BiCRS pathways depend on the specific technology and biomass used but can include carbon-negative routes to waste biomass disposal (carbon that would have otherwise been re-released to the atmosphere), improved air quality when pile burning is avoided, and improved forest health from forest management, as well as the production of fuels that displace fossil fuels and the creation of new, needed industries focused on carbon removal. Technologically advanced BiCRS conversion methods, such as pyrolysis and gasification, as well as newer combustion facilities, are also expected to have improved emissions profiles relative to older combustion facilities (28).

The goal of this chapter is to provide an assessment of promising near-term (deployable at scale by 2030) BiCRS pathways for negative emissions in the United States. Our assessment is based upon conversion technology maturity and economics, waste biomass feedstock type and availability, most-promising regions for sourcing waste biomass (biomass quantities and transportation considerations), and costs/t$CO_2$ of the complete pathway, including costs of biomass collection, transport, conversion, and $CO_2$ storage.

## Current status of BiCRS technologies

The most mature BiCRS conversion processes compatible with woody biomass include combustion, gasification, and pyrolysis integrated with carbon capture and storage (CCS). All three technologies are at large-scale pilot to full commercial-scale for each process unit required. However, in some cases, the process units have not yet been fully integrated in a commercial-scale project. For example, no biomass gasification + CCS projects are commercially deployed in the United States, but biomass gasification and CCS projects *separately* are technologically mature.

Gasification is a high-temperature (>700 °C) processing method that converts carbonaceous materials (coal, biomass) into syngas (a mixture of $CO_2$, CO, and hydrogen) and other solid and liquid compounds. Most current biomass gasification projects produce a liquid fuel, such as ethanol (Enerkem) or jet and diesel fuels (Fulcrum biofuels, Red Rock Biofuels, San Joaquin Renewables), at a scale of 10,000 to 300,000 dry tons of biomass per year without CCS. Pyrolysis is a lower-temperature (500 °C) thermal

processing method used primarily for production of bio-oil from woody biomass. This bio-oil can be used directly for carbon sequestration (Charm, 100 tCO$_2$/y-scale in 2021) or heating (BTG-Green Fuel Nordic Lieksa) and can be converted to conventional liquid or gaseous fuels (Ensyn). Pyrolysis plants typically operate at smaller scales than gasification plants (10,000s vs 100,000s dry tons of biomass/y) and may therefore be more easily placed near the biomass source. However, the smaller scale of pyrolysis and the number of products produced (char, bio-oil, syngas) make it less economically attractive to apply CCS. There is, however, modest carbon storage in the biochar from pyrolysis.

## Technologies/pathways considered for this report

While BiCRS pathways can include a wide range of feedstocks, including high-moisture or gaseous feedstocks, here we focus on BiCRS processes compatible with low-moisture cellulosic/woody biomass because of the large capacity of these biomass resources in the United States. We focus on two specific BiCRS technologies in our analysis: 1) the construction of new biomass gasification facilities integrated
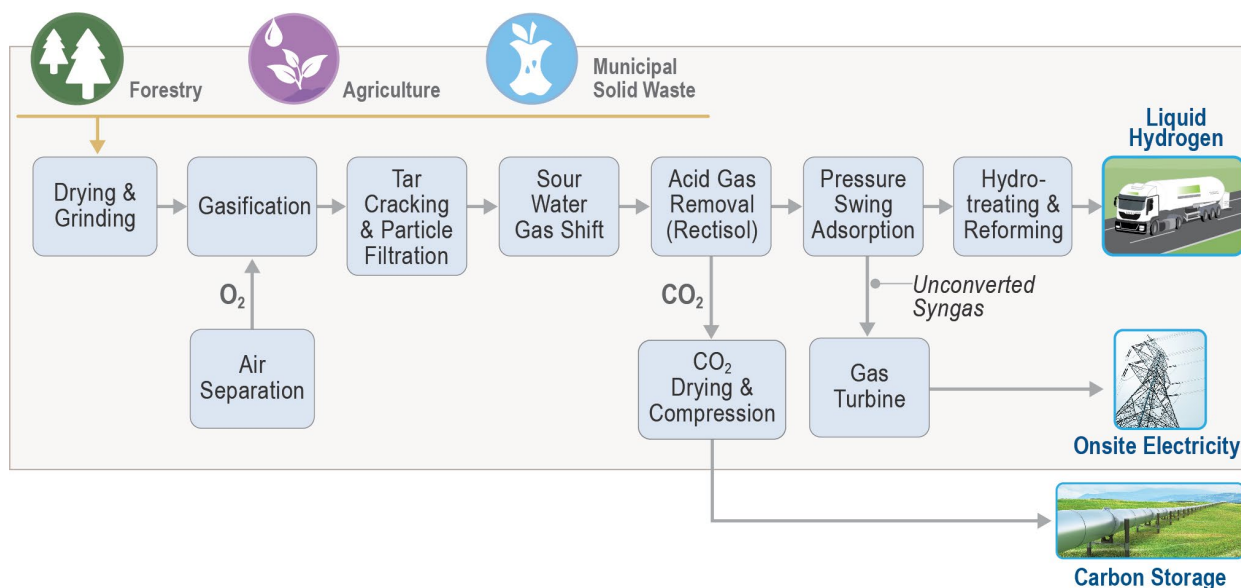


**Figure 2-3.** *Process flow diagram depicting gasification of waste biomass to liquid hydrogen. Our cost analysis in this chapter accounts for each process unit within the system boundary shown.*

with CCS and 2) the retrofit of existing combustion facilities with CCS. For the gasification technology, we focused on gasification of waste biomass to liquid hydrogen with capture of the process CO$_2$ because our previous comparison of this route (*5*) with other thermal biomass conversion technologies and conversion products indicated that gasification to hydrogen is among the lowest cost, highest CO$_2$-yield options for negative emissions from BiCRS. We specifically focus on the production of liquid hydrogen as opposed to compressed hydrogen because liquid hydrogen is a high-value product with mature infrastructure for transport and use. Additionally, production of carbon-negative hydrogen provides an alternative to conventional, natural-gas-derived hydrogen. Note that, although we focus on gasification to liquid hydrogen for our cost assessments, gasification facilities can also be operated to produce renewable natural gas, compressed hydrogen, electricity, and liquid fuels such as ethanol and drop-in diesel.

The second biomass conversion pathway we analyzed (a new analysis for this report) is the retrofit of existing bioenergy plants with CCS; this pathway may provide low-risk, near-term deployment because it leverages existing infrastructure and biomass sources/collection agreements.

Questions of sustainability, additionality, and net negativity for non-waste biomass feedstocks are very complicated. For example, while corn ethanol fermentation off-gas is a concentrated $CO_2$ stream and more straightforward to capture, the overall lifecycle emissions of $CO_2$ is often net positive due to energy input during processing and emissions from agriculture. Similarly, energy crops pose complex questions around land and resource use and are not generally targeted for simple combustion. As more and cheaper renewable electricity becomes available, the addition of carbon capture and storage to generate valuable and robust carbon removal credits is a potential route for outdated combustion
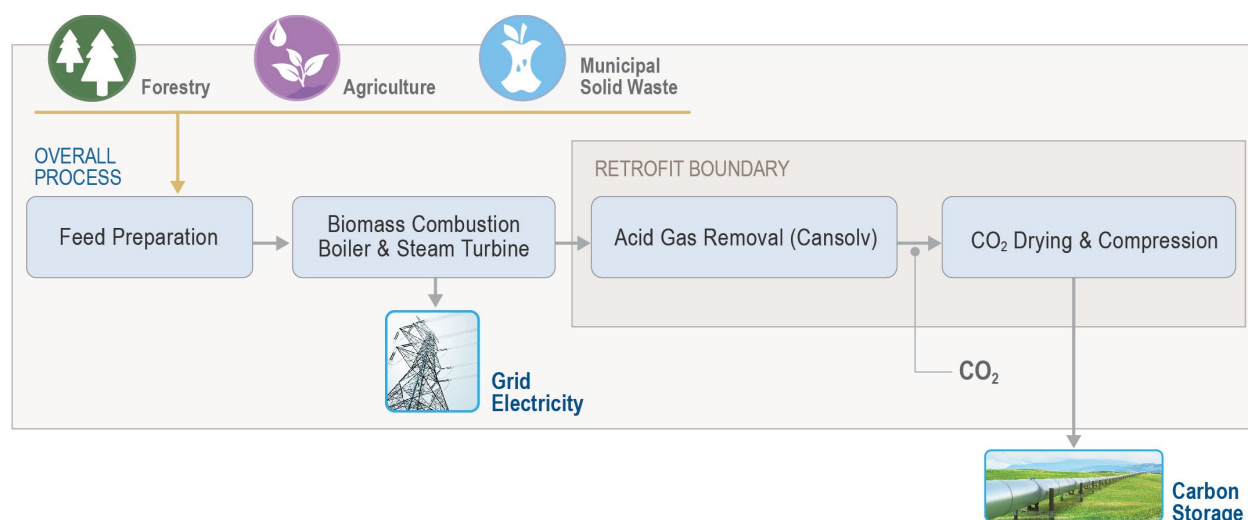


**Figure 2-4.** *Process flow diagram depicting retrofit of existing biomass combustion facilities. Our cost analysis in this chapter accounts for each process unit within the system boundary shown.*

facilities to stay in use and remain economically viable. This pathway has the added benefit of taking advantage of equipment and infrastructure that may otherwise be left unused. Oxy-fuel combustion of biomass fuels or fuel blends is a more energy-efficient and very promising approach to carbon-negative bioenergy. However, new build oxy-fuel plants cannot compete economically with new build biomass gasification to hydrogen, and retrofitting existing biomass combustion facilities for oxy-fuel combustion with CCS would require very significant plant retooling (*29–32*). For these reasons, we chose to focus on simple retrofits of existing bioenergy combustion plants emitting biogenic $CO_2$ with mature absorber CCS technology and without alteration of the combustion process. The economics of new-build bioenergy facilities are challenging compared to more complex biomass conversion processes like gasification, but the reduced capital investment to retrofit one of the hundreds of existing plants in the United States makes this a promising avenue for carbon removal with geologic sequestration and low-carbon electricity. For simplicity, we refer to our two pathways of interest in this chapter as *gasification* and *retrofit*.

# BICRS ANALYSIS OVERVIEW

The gasification pathway relies on new construction; therefore, we assessed the most-promising U.S. counties for construction of new gasification plants integrated with carbon capture. Plant location considerations include locating the source and quantity of waste biomass, the mode of biomass transport, and the site of $CO_2$ storage. In the following sections, we outline our assessment of most-promising geologic storage locations, regional- and county-level biomass availability, and biomass transport modes and costs. The inset shows a high-level overview of the elements we considered in our gasification analysis, leading to an estimate of full-system cost in $/t$CO_2$, including biomass collection, biomass transport, gasification to hydrogen, and geologic storage of the process $CO_2$.

**Gasification Pathway Analysis Elements**

County level biomass availability and cost ($/ton biomass)

Identify and map geologic storage counties

Calculation of minimal cost per county of biomass transport to storage county by road or rail ($/ton biomass*km)(km)

Biomass gasification with carbon capture and geologic storage costs ($/ton $CO_2$)

The retrofit pathway considers the addition of conventional solvent-based industrial carbon capture to existing biomass combustion facilities in the United States using Environmental Protection Agency (EPA) Emissions & Generation Resource Integrated Database (eGRID) data. As shown in the analysis elements inset, we selected plants for potential retrofit based on proximity to geologic storage, feedstock type, and plant size. We estimated the costs for retrofit and associated $CO_2$ removal using the Integrated Environmental Control Model developed by Carnegie Mellon University (CMU) and the National Energy Technology Laboratory (NETL), with primary variables of fuel type, plant size, capacity factor, and carbon-capture solvent-type. We highlight several representative examples of existing facilities and cost curves for total capital requirement and overall cost of removal per ton of $CO_2$ for a range of plant sizes relevant to feasible retrofit opportunities.

**Retrofit Pathway Analysis Elements**

Down-select existing biomass combustion facilities for retrofit based on size and feedstock

Highlight regions of opportunity: largest existing facilities and geologic storage

Calculation of retrofit costs and resulting cost of capture and storage for example facilities

Generate general cost curves for carbon capture retrofit based on plant size and capture potential

## Gasification: Assessment of promising biomass regions and $CO_2$ costs

### *Waste biomass availability and cost*

We obtained data on U.S. waste biomass availability from the U.S. Department of Energy (DOE) 2016 Billion-Ton Report (BTR) (*33*). The report contains estimates of current and potential county-level biomass production and density for both wastes and bioenergy crops. Because we focused on 2030 deployment opportunities, we used the most conservative assessments of biomass availability. We used 2017 data on availability, without any projections on future availability, and only included the waste

portion of currently produced biomass (outside of any current use). The broad categories included in our analysis were a subset of 1) agricultural wastes, including corn stover; 2) wastes from the forestry (timber only) industry outside of any current use, including mill wastes; and 3) select MSW. We included only three subtypes of biogenic MSW: construction and demolition, paper and paperboard, and wood waste. We chose the waste subtypes in this analysis based on suitability for gasification (less than 15% moisture content) and on the stipulation that the waste be composed of only biogenic carbon.

Table 1 shows the 2017 estimates of U.S. waste biomass we used in our analysis, as well as the assumed collection costs of the waste as stipulated by BTR. The report database provides availability of biomass according to biomass cost. Some agricultural wastes, such as rice straw and cotton residue, are only available in the BTR database at $60/ton. Since these are produced in significant quantity, and our intention is to show the range of costs in the U.S., we included these wastes in our assessment. The collection costs and availability listed in the table correspond for all wastes except for paper and paperboard; here we assumed that the maximum paper and paperboard was available at $80/ton but the collection cost is zero because this waste is already collected (34). We want to note that the MSW-derived waste represented in the table is a conservative estimate. Conversations with experts indicate that the resource is likely larger—potentially two-fold—and research is underway to gain clarity on the amount of resource available (34, 35). Furthermore, our conversations with experts indicate that collection costs (in this case, sorting costs) and fates of biogenic MSW are not well understood. Here we chose the conservative estimate of $40/ton for most biogenic MSW in our accounting, but because utilization of otherwise landfilled waste results in an avoidance of tipping fees, the collection cost is likely much lower. However, we identified both the fate and collection/sorting costs for U.S. biogenic waste as a major research need. Finally, we note that we include here biomass from logging residues from timberlands according to USDA Forest Service (USFS) Forest Inventory and Analysis (FIA) data and the Forest Sustainability and Economic Assessment Model (ForSEAM) in BTR. In contrast, the forest biomass assessment in Getting to Neutral[13] used USFS BioSum and Forest Vegetation Simulator to model treatment of 800,000 acres per year of timberland for 1) effective reduction in potential fire mortality, 2) ability to generate positive net revenue (i.e., not lose money), and 3) ability to maximize in-stand carbon. The BTR also reports that harvests of small-diameter whole trees on timberlands, which could mitigate fire risk, could produce an additional 60–70 million tons per year at $60 per dry ton but is excluded from this analysis.

**Table 1-1**. Waste biomass quantities and collection costs used in our analysis.

| Category | Waste | U.S. resource, annual short tons (2017) | Collection cost ($/ton) |
| --- | --- | --- | --- |
| **Agriculture** | Corn stover | 23,905,000 | 40 |
| **Agriculture** | Rice straw | 4,911,000 | 60 |
| **Agriculture** | Cotton residue | 3,733,000 | 60 |
| **Agriculture** | Sugarcane bagasse | 3,610,000 | 40 |

| Category | Waste | U.S. resource, annual short tons (2017) | Collection cost ($/ton) |
|---|---|---|---|
| **Agriculture** | Non-citrus residues | 2,520,000 | 40 |
| **Agriculture** | Cotton gin trash | 1,739,000 | 40 |
| **Agriculture** | Tree nut residues | 1,509,000 | 40 |
| **Agriculture** | Citrus residues | 1,499,000 | 40 |
| **Agriculture** | Rice hulls | 1,375,000 | 40 |
| **Agriculture** | Sugarcane trash | 503,000 | 40 |
| **Forestry** | Other forest residue | 12,246,000 | 40 |
| **Forestry** | Hardwood residue (upland and lowland) | 6,862,000 | 40 |
| **Forestry** | Softwood residue (natural and planted) | 6,803,000 | 40 |
| **Forestry** | Mixed-wood residue | 4,242,000 | 40 |
| **Forestry** | Secondary mill residue | 4,139,000 | 40 |
| **Forestry** | Primary mill residue | 486,000 | 40 |
| **MSW** | Construction and demolition waste | 22,176,000 | 40 |
| **MSW** | Paper and paperboard | 16,055,000 | 0 |
| **MSW** | MSW wood | 5,019,000 | 40 |
| **Total** | | **123,330,000** | |

*Geologic storage regions*

We used data on $CO_2$ storage regions from both the National Carbon Sequestration Database (NATCARB) and the United States Geological Survey (USGS), as well as $CO_2$ pipeline information from Baik et al. (*36*). We used Baik et al.'s injectivity cutoff of 250,000 $tCO_2$/y/well as our threshold for regions most economical (due to economies of scale) for injecting $CO_2$. We only considered these high injectivity regions in our calculations of biomass transport costs based upon distance between biomass locations and storage regions. The $CO_2$ storage regions depicted in Figure 2-5 are vast, even when applying an injectivity threshold, reducing the range of calculated transportation costs beyond what is likely realistic. Higher-resolution mapping of most promising near-term geologic storage locations, beyond injectivity, is needed but must also include other factors like population density, proximity to other natural resources,

and other considerations (*37*) to determine more accurate biomass transportation costs and help identify near-term project sites.
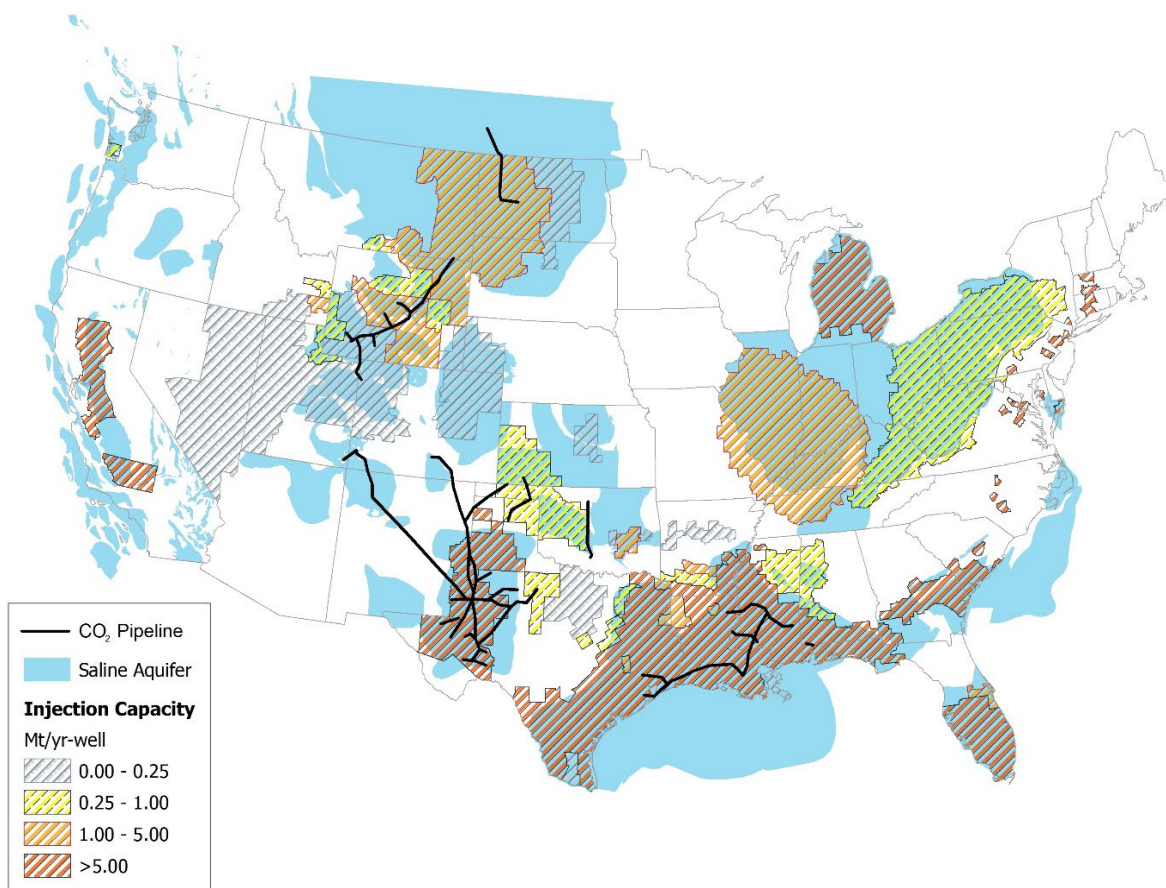


**Figure 2-5.** *Overlay of U.S. geologic storage regions according to NATCARB and USGS. For our analysis of biomass transportation costs, we assumed most-promising storage regions corresponded to injectivity greater than 250,000 tons/y/well according to analysis of USGS data from Baik et. al*

### Biomass transport assumptions

Gasification to hydrogen has a high $CO_2$ "storage factor" of about 1.6 $tCO_2$ per ton biomass. Our previous analysis indicated that, for conversion technologies with high storage factors, *transporting biomass by truck or rail is less expensive than transporting $CO_2$ by the same mode* (*38*). Therefore, we assume here that the gasification pathway requires sourcing biomass and transporting it via the minimum-cost route (truck or rail) to a new gasification facility located within a $CO_2$ storage region. For biomass transport costs, we used $0.101/ton/km for truck and $0.044/ton/km for rail (*38*).

## GIS methodology for calculating transport costs

Network analysis is a methodology to understand and expedite movement along a set of pathways. We applied network analysis in this instance to identify counties or groups of counties designated as "regions of opportunity" where transporting large amounts of biomass for processing via road or rail to

$CO_2$ storage areas would be low cost. The Closest Facility algorithm in the ESRI ArcGIS Desktop software with the Network Analyst extension provided the means to accomplish this task.

Prior to implementing this analysis, numerous pre-processing steps were necessary. The required macro-scale steps included identifying or calculating the following for both road and rail:

- Storage areas or existing $CO_2$ pipelines to which the biomass would be transported
- Networks for road and rail lines
- "Incident" points at each county to serve as the source location the biomass
- "Facility" points in those storage areas to serve as the destination for the biomass
- Closest facility analysis
- Minimum cost of transport for all routes

We note that all datasets for this analysis were projected into the coordinate system NAD83 Contiguous USA Albers prior to any processing.

### Georeferencing of designated storage areas

$CO_2$ storage areas were designated as those with injection capacity greater than 0.25 Mt/y/well or as a $CO_2$ pipeline, as shown in Figure 2-5, with georeferenced data from Baik et al (2018) (*36*). Straight heads-up digitization would not have been as accurate as desired due to the slight distortion of the image during georeferencing. Fortunately, the shapes of the counties that bound the regions of interest were quite discernable, which allowed us to select the counties and export them to a new feature class to obtain a satisfactory boundary in the correct coordinate projection. These groups of counties were then dissolved into larger features corresponding to the different injection capacities depicted in the original figure.

We also designated $CO_2$ pipelines as "storage areas" and thus digitized them from this georeferenced figure as well. The georeferencing to digitize the $CO_2$ pipelines was done separately from the injection capacity areas so that pipelines could be digitized with the maximum accuracy possible using this method. We also took care during digitization from the georeferenced image to account for the location of each pipeline within each county so that the location accuracy could be improved further, even if it did not quite match the georeferenced imagery.

### Road and rail networks

We used the Railways, HIFLD Open (accessed October 2019) dataset to develop the rail network and the Roads, MapCruzin.com (2020) dataset to develop the road network. We set up a dedicated geodatabase for both road and rail and, within each geodatabase, we created a dedicated feature dataset for a network dataset. We imported the corresponding lines' feature class into each feature dataset. Prior to creating a network dataset, we erased from each dataset the road and rail features within the storage areas described above, truncating them at the boundary of the storage areas. Then, to reduce the number of routes that would need to be calculated and thus expedite processing, we dissolved the two datasets to consolidate segmented features into one. Finally, we created and built a new network dataset for both road and rail with this subset of features using the tool provided by the software.

### *"Incident" points at each county to serve as source location for biomass*

Closest facility analysis requires a point, or many points, to serve as an incident location. In this instance, the intersection of the county boundaries and rail lines served as incident locations. Note that many counties in the United States are not covered by a rail line from the HIFLD Open dataset; however, other counties may be crossed several times by rail lines. Thus, certain counties may have more than one incident (or source) location, while others have none. Due to the large number of incident locations, we implemented a manual country-wide quality assurance (QA)/quality control (QC) process to remove incident points that either could not connect to a storage location (for example on an island with no bridge in the dataset to the mainland) or would clearly not provide a minimum path to a storage area (for example, when a road line crisscrossed a county boundary many times in a very short distance). Once we completed the QA/QC process for incidents, we used the "Add Locations" tool to populate these incidents to the network dataset.

### *"Facility" points in storage areas to serve as biomass destination*

A second input to the closest facility analysis is a destination point called a facility. In this instance, the intersection of the $CO_2$ storage areas (boundary of the injection capacity areas with 0.25 Mt/y/well or existing $CO_2$ pipeline) and the rail lines (or roads) served as the facilities. To expedite the closest facility analysis, we performed the same manual QA/QC check of facility points to reduce the number of route calculations that were needed and thus reduce processing time. Once we completed the QA/QC process for facilities, we used the "Add Locations" tool, this time to populate facilities to the network dataset.

### *Closest facility analysis*

With the network built and incidents and facilities added, the next step implemented the Closest Facility algorithm. We entered each parameter into the algorithm and set the impedance attribute to length (kilometers).

The Closest Facility algorithm calculates a "route" layer with the distance between each incident and its closest facility. Due to the existence of multiple incidents for a given county in both the case of roads and rail, in some instances multiple routes were calculated to storage locations for a given county. Thus, to identify just the route with the minimum distance to storage for each county required additional processing, which we did outside of GIS.

### *Calculating minimum cost of transport for all routes*

For each rail and road, we exported the "route" layer attribute table into Excel. We created a pivot table with counties listed by Federal Information Processing Standards (FIPS) code and the minimum route distance. Next, we added a new field for cost/ton/km (rail = $0.044, road = $0.101) and calculated a field for cost/ton by multiplying the minimum route distance in km by the cost/ton/km.

With cost/ton for each route, we used data from the U.S. DOE 2016 BTR study to calculate total production and production density for three categories of biomass: agriculture, included with $40 biomass availability, except for cotton residue and rice straw, which were included with $60 availability; MSW, included with biomass availability at the county level of $80; and forestry, included with $40

biomass availability. Using a pivot table, we summed the total biomass and biomass production density for each county.

We imported the pivot tables for both biomass amounts and transport and joined each with the county data layer by the unique FIPS code for each county, which were then incorporated into the visualizations for the project on a bivariate scale.

### *Assessing processing and geologic storage costs*

To calculate processing costs for gasification, we relied on the technoeconomic analysis in Larson et al., whose analysis was based upon switchgrass, a good proxy for the wastes we considered here (*39, 40*). Our process assumptions were as follows: selling price for liquid hydrogen – $4.00/kg; plant lifetime – 20 years; discount rate – 12.5%; and plant uptime/capacity factor – 90% (*41–43*). Given that the literature analysis was based upon a large gasification plant size of 4500 tons of biomass per day, we estimated the costs of smaller plant sizes that are more realistic for near-term deployment by scaling the process unit capital costs based on the Quality Guidelines for Capital Cost Scaling Methodology from NETL (*44*), using reference process unit sizes and capital costs from the literature.

From our process model based upon the assumptions above, we were able to simplify the cost calculation for a given plant size to the following: biomass collection + transport cost ($/ton biomass) * (1 ton biomass / 1.65 $tCO_2$) + a constant for each plant size. These constants are 3.28, -32, and -52 for 1000, 2000, and 4500 ton/day–sized biomass gasification plants, respectively. We assumed a fixed geologic storage cost of $10/$tCO_2$ based upon analysis by NETL (*45*).

## Retrofit: Identifying facilities of interest and modeling costs

### *Evaluating existing combustion facilities*

We obtained data on existing combustion facilities and their associated fuels and emissions through the U.S. EPA's eGRID (*46*). These data include information on fuel type, plant nameplate capacity and capacity factor, net generation, and emissions overall and from combusted biomass. The United States has hundreds of existing biomass combustion facilities. The biogenic feedstocks for these facilities include agricultural waste (Ag), black liquor (BLQ), landfill gas (LFG), MSW, and wood waste.

Some existing biomass combustion facilities are co-fired with other non-biomass fuels, most commonly coal or natural gas. Co-fired facilities are not desirable for retrofit as co-firing reduces the portion of the captured and removed $CO_2$ emissions that is biogenic, and the resulting capital and operating costs for carbon removal are greater. Theoretically, a cost associated with positive emissions would reduce this effect, and larger co-fired facilities could be promising retrofits with mixed $CO_2$ emission abatement and removal. BLQ and LFG combustion facilities are also generally not favorable for retrofit for a range of factors including small scale (LFG), impurities, and commonly being co-fired with fossil fuels (*47*). Thus, in this study, we focus only on existing facilities burning Ag, MSW, and wood waste. Importantly, MSW is not entirely biogenic, depending on the mixture, sourcing, and components, and in this study we assumed that $CO_2$ captured from MSW combustion is 50% biogenic (*46*). This assumption has a similar affect to co-firing as it increases the cost of removal because only a portion of the $CO_2$ captured may be counted as removal. Sourcing particular subgroups of MSW that are more or entirely biogenic as

combustion feedstocks may be possible, but we did not probe alternative sourcing or fuel adjustments as those scenarios are highly case- and region-specific and may be restricted due to standing contracts.

Most existing biomass combustion facilities have relatively small nameplate generation capacities; 28 facilities have nameplate capacities ≥100 MW, and only 3 facilities have nameplate capacities ≥200 MW. These small scales are economically unfavorable for retrofit, especially in comparison to the potential for carbon capture at fossil fuel combustion plants, which are often significantly larger. Other limiting factors include very low facility uptime (capacity factor) and significant co-firing with fossil fuel, often coal. Still, these facilities, especially the largest among them, have potential as near-term and relatively low-capital-cost opportunities for carbon removal through retrofits with conventional flue gas carbon capture technologies.

## *Modeling carbon capture retrofit cost in the Integrated Environmental Control Model (IECM)*

We used the Integrated Environmental Control Model (IECM), a tool developed by CMU and NETL, to calculate performance, emissions, and economics of existing and theoretical combustion facilities (*48*). IECM has straightforward, established methods for calculating economic details of carbon capture retrofits to biomass combustion facilities. In general, estimating retrofit costs is challenging at very small scales, and most scaling factors and models available in the literature target plants with nameplate capacities in the hundreds of megawatts (MW) (e.g., 4000–7000 $tCO_2$/d). This issue complicated our analysis as only a small fraction of the existing facilities have nameplate capacities ≥100 MW (~1000 $tCO_2$/d). The IECM modeling tool features a rough lower bound of 100 MW gross generation, below which the scaling models may not be accurate. For this reason, the minimum nameplate capacity (i.e., net generation) plant we considered was 75 MW.

### Overview of the Integrated Environmental Control Model (IECM)

CMU and NETL's IECM tool is a robust and flexible tool for modeling the operation and economics of solid fuel (pulverized coal), natural gas combined cycle, and internal gasification combined cycle plants and has been used in many published studies (*19*, *49–52*). IECM can readily model solid biomass fuels using defined custom fuel compositions and heating values (*53–56*) and can also model a variety of post-combustion controls, including $CO_2$ capture. IECM users can specify a vast array of system parameters, including key performance factors like gross output, boiler efficiency, and capacity factor, as well as economic factors such as regulated emissions constraints, financing, discount rate, charge factors, federal and local taxes, and fuel costs. Unless otherwise specified, we used default recommended values for solid fuel combustion plant operation; some parameters are specified in this section.

### Setup and workflow for retrofit cost evaluation

Analysis of $CO_2$-capture retrofit economics is a specified workflow described by the creators of IECM, and we generally followed their recommended approach, which establishes a baseline plant with fully amortized equipment and subsequent addition of a new carbon-capture unit. The gross generation is then reduced to decrease the fuel rate to the pre-retrofit level. The baseline plant includes only $NO_x$ control with hot-side selective catalytic reduction (SCR) in its post-combustion controls; other emissions, including mercury, particulates, and $SO_2$, are not evaluated.

## Key assumptions for retrofit modeling

Baseline breakeven operation was a critical assumption used in cost evaluation of the biomass combustion facility retrofits. With this assumption and with an unchanged fuel rate, we assumed that the fuel being burned had an effective cost of zero, much like we assumed the existing equipment to be fully amortized. For specific cases in which fuel rate was increased, we accounted for the cost of any additional fuel above the baseline plant rate. We will discuss these cases further in the section describing the retrofit scenarios we examined.

We considered retrofits utilizing conventional absorber–stripper systems with two different liquid solvents: monoethanolamine (MEA), an older and more mature technology, and Cansolv, a more modern solvent system from Shell. We did not consider a third solvent option available in IECM, Fluor FG; its economics and efficiency were similar to those of Cansolv, so we arbitrarily chose just one of them for simplicity. Table 2-2 shows some key parameters used in the model.

**Table 2-2.** Key system and economic parameters used in all IECM retrofit models.

| Parameter | Value |
| --- | --- |
| Capture efficiency | 90% |
| Boiler type | Sub-critical |
| Capital discount rate | 10% |
| Fixed charge factor | 13.4% |
| Post-retrofit plant life | 20 years |
| Construction time | 3 years |
| Electricity sale price | $60/MWh |

Models of specific plants from the eGRID database of existing combustion facilities utilized the listed nameplate capacity (set to be the baseline plant net generation) and capacity factor. The theoretical plants used in our cost curve models all assumed a capacity factor of 85% across a range of nameplate capacities. These theoretical facilities also used regional values from the Southeast United States, as this region has the greatest density of potential retrofits. For all cases, we assumed the storage cost of $CO_2$ to be $10/tCO_2$.

## Model outputs and analysis

The model calculates a cost of captured $CO_2$ by comparing the internal cost of electricity, or minimum selling price, of the baseline plant to that of the retrofitted plant. This comparison incorporates all economic factors, including the annualized capital and operating expenses. Because we kept the fuel rate constant, the addition of carbon capture operations (absorber, stripper, etc.) significantly reduced the net electrical output of the plant. We did not consider the addition of new boilers or increased

generation by burning additional natural gas. In one scenario listed below, we considered increasing capacity factor with additional biomass fuel.

We added a loss in revenue from reduced generation (using the electricity sale price in Table 2-2) to the internally calculated cost of $CO_2$ capture, and we recorded the absolute and fractional loss in net generation with retrofit. Finally, we noted the effective total capital requirement for the retrofits, including annualized capital and operational and maintenance costs. Additional details and example cost breakdown may be found in the appendix.

## *Retrofit scenarios examined*

### Case studies of existing facilities

We selected several of the most-promising existing facilities from the eGRID data as case studies for retrofit and modeled them with their reported fuels, size, and capacity factors. We selected one facility of each fuel type—wood waste, agricultural waste, and MSW (Table 2-3)—for consideration based on scale and proximity to geologic storage regions.

**Table 2-3.** Retrofit case study biomass combustion facilities.

| Plant name | Fuel | County, State | Nameplate capacity (MW) | Capacity factor (%) | Annual biogenic $CO_2$ emissions (Mt) |
|---|---|---|---|---|---|
| Clewiston Sugar House | Agricultural waste | Hendry, FL | 90 | 32 | 2.1 |
| Covanta Fairfax Energy | Municipal solid waste | Covanta, VA | 124 | 61 | 0.54 |
| Deerhaven Renewable Energy | Wood waste | Alachua, FL | 116 | 56 | 0.71 |

We considered two scenarios for each case study. In Scenario 1, we kept fuel use and capacity factor constant, leading to a significant decrease in net generation. In Scenario 2, we increased capacity factor to remove any net generation losses associated with the energy required for the operation of the carbon-capture unit. This adjustment increased the required biomass fuel, and we accounted for the cost of this additional fuel according to the weighted average cost for that fuel in that specific county using data from BTR and relevant biomass types shown in Table 2-1. This second scenario accounts for contracts governing energy generation in which the original net generation may be maintained by increasing fuel supply. We selected this approach as we assumed a plant was more likely limited by generation demand and contracts than by fuel supply; however, this assumption may not always be the case.

### Cost curves for generic facilities

In addition to the case studies of specific plants, we generated generic cost curves for carbon capture retrofits of theoretical biomass combustion facilities. We evaluated overall capture cost ($/tCO_2$) and effective total capital. As with Scenario 1 in the case studies, we kept the fuel rate of each baseline plant

constant, resulting in an associated loss in net generation and electricity sale revenue. We modeled biomass combustion plants with nameplate capacities in the range of 75–300 MW for each of the three target fuel types.

## RESULTS

### Waste biomass availability in the United States

Our major finding from our waste biomass assessment is that approximately 123 million tons of biomass are available per year with the constraints described the Analysis section (Table 2-1). This amount corresponds to around 200 million tons of $CO_2$ per year that could be sequestered through the biomass gasification + CCS pathway. The most significant biomass sources among the three waste categories are 1) corn stover from agriculture, 2) construction and demolition waste from MSW and 3) other forest residue from forestry operations.

Various considerations will influence the choice of waste biomass type (agriculture, forestry, MSW) and subtypes (corn stover, sugarcane trash, etc.) for gasification, including feedstock homogeneity, certainty, seasonal availability, and others. We did not extensively research these considerations. Broadly, industrial entities that would put forward the capital to build such a plant will be interested in long-term, compositionally homogeneous, reliable sources of biomass. Other factors that we did not consider, such as silica content and economics of biomass pre-processing, may also exist. We were not able to quantify these considerations, but we did include estimates of collection costs from BTR (Table 2-4). Agricultural wastes in our dataset included some subtypes with availability at $60/ton; specifically, we chose to include cotton residue and rice straw, which are only available at higher collection cost, to represent the range of costs, with most selected subtypes at $40/ton. Therefore, agricultural wastes were the highest cost source of biomass in our analysis, but this value is highly dependent on the choice of waste. MSW is the lowest cost source of biomass in our analysis, due to avoidance of tipping fees and our assumption of zero collection cost for paperboard. However, current use (e.g., recycling) and costs of sorting and handling non-recycled paperboard requires more research.

**Table 2-4**. Weighted average collection costs per waste biomass type.

|  | Million dry tons/yr | Wt. avg. collection cost ($/ton) |
|---|---|---|
| **Agriculture** | 45.3 | 43.8 |
| **Forestry** | 34.8 | 40 |
| **MSW** | 43.2 | 25.2 |
| **Total** | 123.3 | 36.2 |
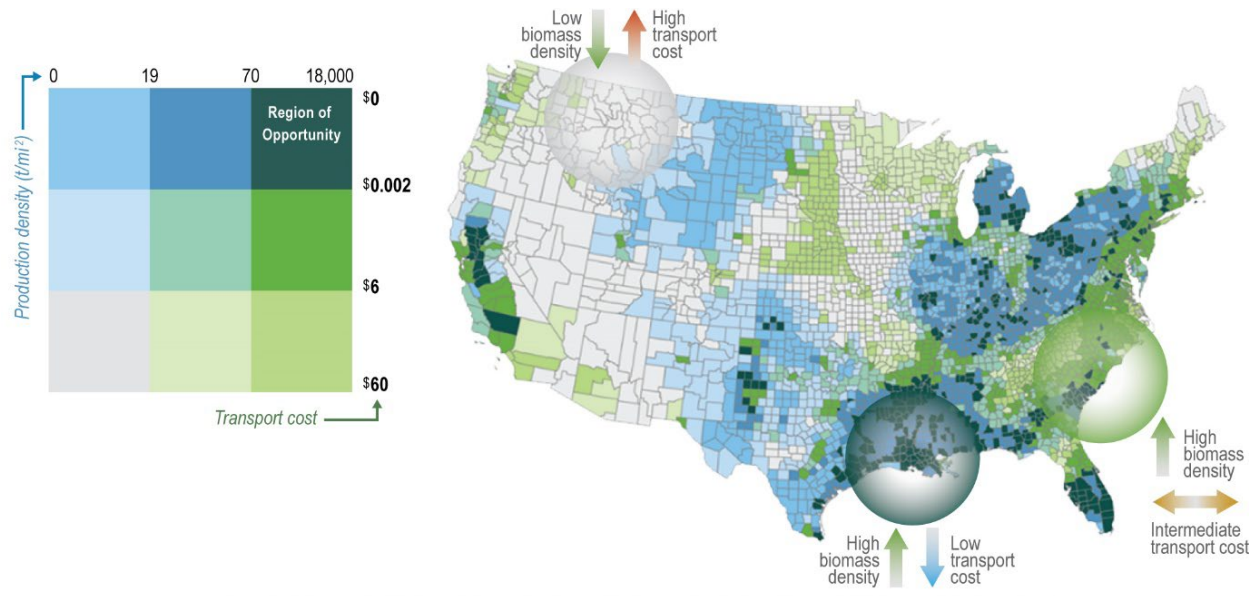
**Figure 2-6.** *Regions of highest opportunity for building new gasification plants in the United States. Dark green counties reflect U.S. counties with roughly the top 1/3 highest biomass density and lowest biomass transportation costs. Biomass density data are from 2017 estimates. Regions highlighted by dark green, light green, and grey circles are examples and are intended to explain the color coding in the legend.*

With the goal of identifying the most-promising near-term U.S. regions and counties for installation of gasification plants, we overlayed the storage regions (including $CO_2$ pipelines) shown in Figure 2-1 and waste biomass density, and used data on road and rail lines, as well as biomass transport costs from our own prior analysis (*38*), to estimate transport costs from each county to the nearest storage site. We used county-level estimates of waste biomass availability to map distribution across the United States according to type. The data are available both as a biomass density and as annual dry tons. Figure 2-5 shows our map of the most promising regions in the United States for locating biomass gasification plants. The most promising regions (highest biomass density and lowest transport cost) are along the Gulf Coast, Central California, Southern Florida, and some counties in the upper Midwest and Northeast. Note that this map only accounts for biomass transport costs, not collection costs. The counties with the most intense shades of green reflect U.S. counties with the top 1/3 highest biomass densities, corresponding to greater than 70 dry tons of biomass per square mile per year. We regard these counties as favorable waste biomass source counties for gasification. Biomass at these densities would require collection from within a 40-mile radius to support one 1000-ton-biomass-per-day gasification facility (which we assumed in process cost calculations). We did not account for the annual cost or capital costs of biomass storage in our calculations. For resources only available seasonally—e.g., agricultural wastes—the gasification facility will need to store up to 2/3 of the annual supply. The least expensive biomass storage option is on-ground unprotected at about $1/ton of biomass. (However, leaving biomass unprotected can lead to annual losses of about 6%.) The most expensive storage option is indoors, estimated at about $25/ton, with very little loss of biomass (*57*). This range of biomass storage costs would add about $4–18/$tCO_2$ for a 1000-ton/day gasification plant. The dark green regions reflect counties with the most favorable circumstances for supporting a gasification plant: high biomass

density and minimal transport costs. In contrast, the gray regions have the lowest waste biomass density and highest transport costs and are currently the least favorable for supporting a gasification plant.

## Gasification process costs

In Figure 2-7, we show the sensitivity of cost/tCO$_2$ to plant size and to weighted average biomass collection costs per biomass type (agriculture, forestry, or MSW). The greatest cost driver is plant size: the smallest plant—1000 tons/day—cost $25 or $37/ton for biomass + conversion costs from MSW and agriculture wastes, respectively. The largest plant—4500 tons/day—is projected to generate a profit per ton of CO$_2$, achieved through lower processing costs (economies of scale) and the sale of liquid hydrogen. We include a large scale 4500 ton/day gasification plant as a point of comparison in our analysis because 1)
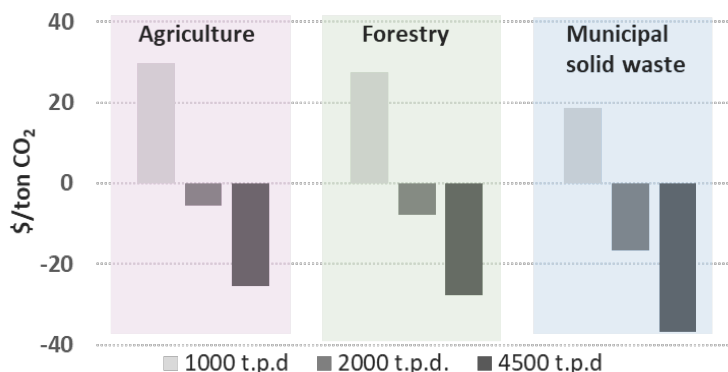


**Figure 2-7.** *Calculated cost/tCO$_2$ for gasification + CCS for agricultural, forestry, and MSW feedstocks. We find that costs are most sensitive to gasification plant size, leading to cost/tCO$_2$ ranging from $-29 to 37/ton (excluding incentives, transport, and geologic storage costs). Negative costs reflect revenue per ton CO$_2$ from the sale of liquefied hydrogen. t.p.d. = tons/day.*

this is a routine size built for gasification of fossil resources such as coal, 2) this is the base case size in Larson et al., from which we drew our processing costs, and 3) we identified a handful of U.S. counties that could support a plant of this size. The profits for the 4500 ton/day plant are $18 and $29/tCO$_2$ from agricultural and MSW, respectively. Table 2-5 shows the capital costs we calculated for the same range plant sizes. The capital required for new-build gasification plants presents a significant risk to implementation, with the smallest 1000 tons/day plant requiring on the order of $500M to construct. For the subsequent cost calculations, we focused only on the smallest plant size as being the most likely to be implemented by 2030. Our rationale for analyzing the 1000-ton/day plant scale is as follows: to our knowledge, no biomass gasification with carbon capture plants are in operation today. However, several projects are nearing deployment at about 500–1000 tons/day to flexibly produce various fuels from biomass gasification, with plans to pursue carbon capture in the future. Additionally, SunGas, a subsidiary of the Gas Technology Institute, has prepared engineering designs for a 1000-ton/day biomass gasifier (*28*), which several California developers are planning to use.

**Table 2-5**. Negative emissions potential and capital costs associated with gasification plant size.

| Gasification plant size (tons biomass/day) | Negative emissions potential (tCO$_2$/y) | Capital cost (M$ 2018 USD) |
|:---:|:---:|:---:|
| **1000** | 525,600 | 541 |
| **2000** | 1,051,200 | 872 |
| **4500** | 2,628,000 | 1700 |
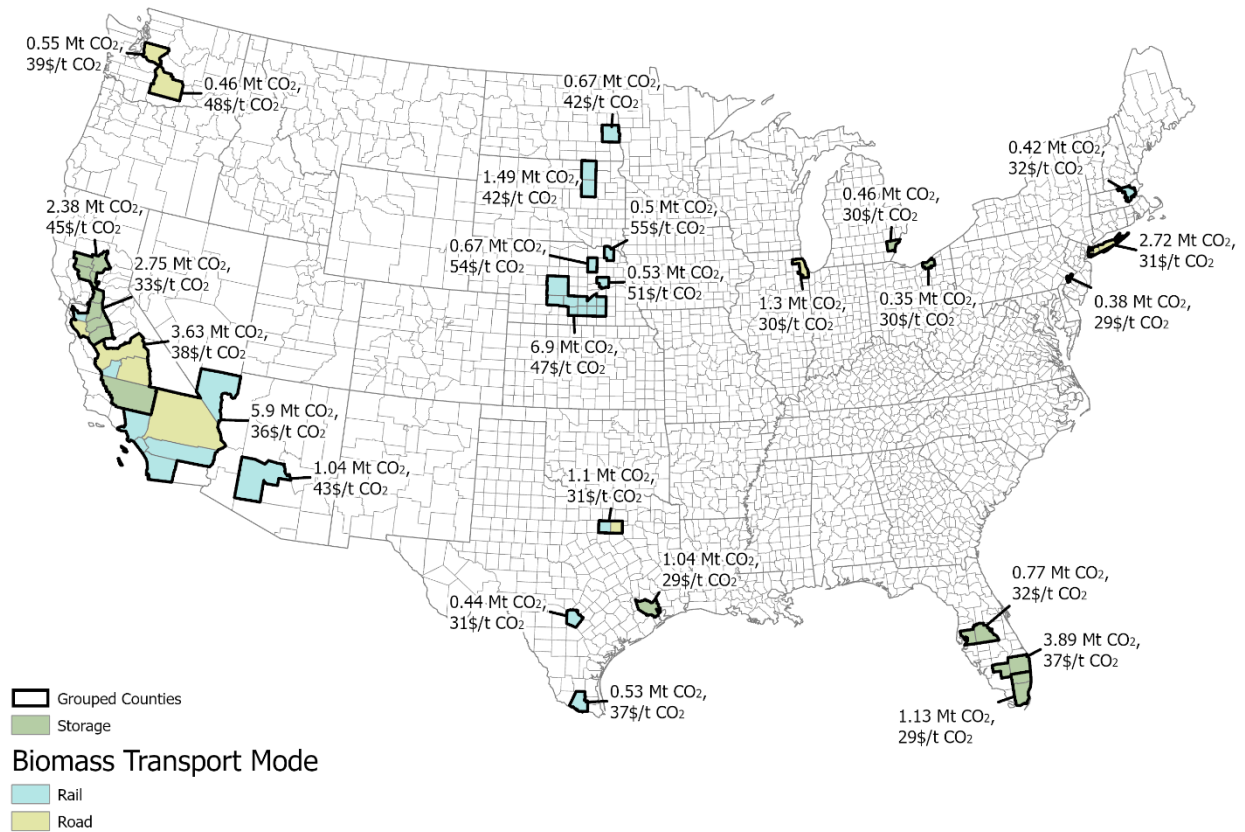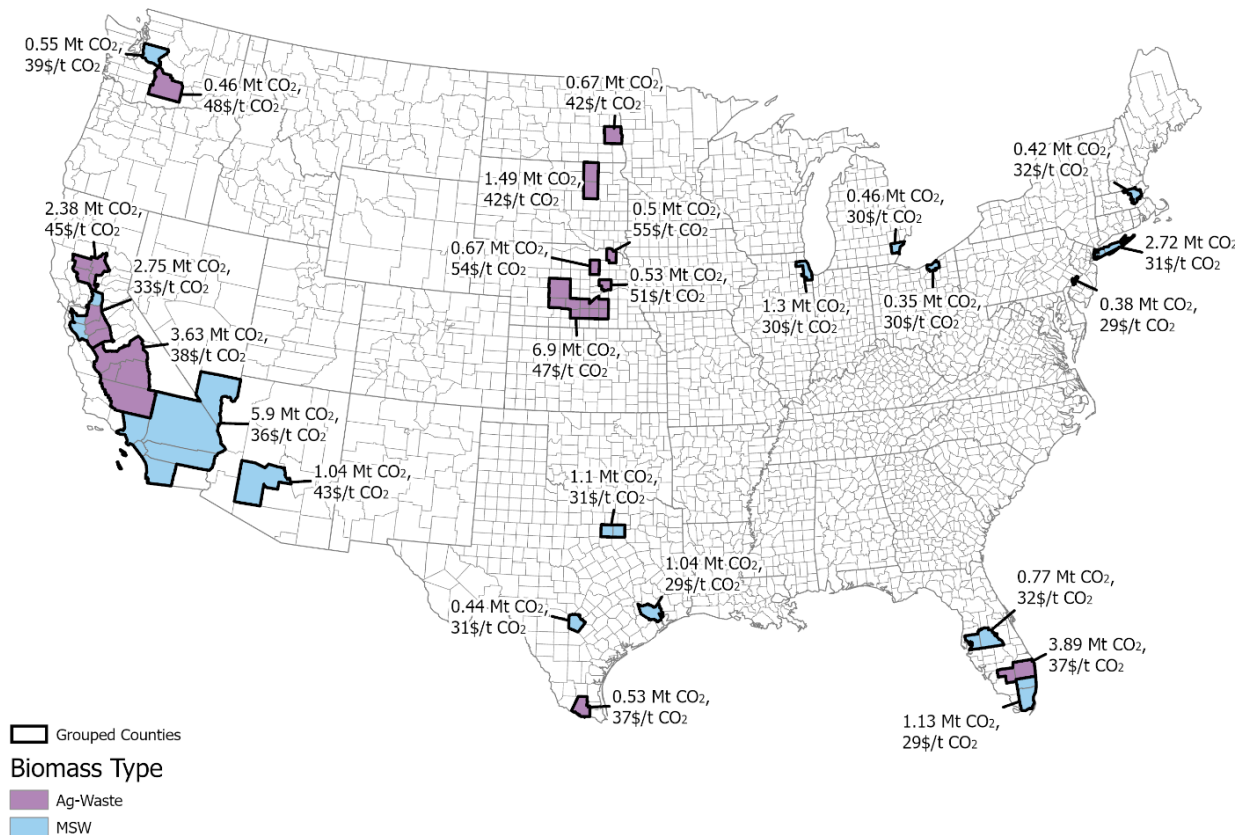
**Figure 2-8.** *For each county or grouped counties indicated by bold outlines in the U.S. maps above, we indicate 1) annual Mt of $CO_2$ potential through gasification (to hydrogen) of waste biomass available within the county/grouped counties and 2) system cost per ton $CO_2$ for a 1000 ton per day biomass gasification plant. The system cost calculation is described in the Analysis section and includes 1) the weighted average biomass collection costs for the specific biomass in each county/region, 2) biomass transportation costs via road or rail (by shortest distance to storage region), 3) processing costs for gasification of waste biomass to liquefied hydrogen with carbon capture for a 1000 ton biomass per day plant scale, and 4) geologic $CO_2$ storage costs. Several of the regions/counties can support multiple plants of this size or could support a larger plant. Top map: County colors reflect the lowest cost biomass transportation mode. The "storage" transport mode reflects counties that can also provide >250,000 $tCO_2$/y geologic storage, so we assumed biomass transport cost is zero. Bottom: County colors reflect the predominant type of biomass in the county.*

We evaluated system-level costs for select U.S. counties based on annual biomass production; we found 60 U.S. counties that could each supply at least 200,000 tons/y of waste biomass, the approximate quantity needed to support a small gasification plant. Figure 2-8 (top) shows system-level costs, including weighted average biomass collection costs for the county, gasification to liquid hydrogen + carbon capture cost for a 1000-ton/day plant, calculated biomass transport costs by road or rail (if applicable), and $CO_2$ storage costs. We want to acknowledge that these counties do not necessarily reflect the "regions of opportunity" depicted in Figure 5 but rather give a snapshot of cost/ton, predominant biomass type, and preferred biomass transport mode for the highest producing counties. Total county production as a metric skews the results toward counties with larger geographic areas.

Therefore, only MSW and agricultural counties are reflected here, as the counties with predominantly forestry biomass are smaller in area. Nevertheless, we anticipate that these forestry counties with high biomass density shown in the "regions of opportunity map"—with biomass collection costs of around $40/dry ton of biomass—will align well with the costs from the agricultural areas in these regions, at around $30/tCO_2. Overall, in this sampling of counties across the United States, total system costs range from $26/tCO_2 in low biomass cost/low transport cost counties to around $50/tCO_2 in high biomass cost and/or high transport cost counties. Figure 2-8 (bottom) also shows the same counties and county costs but indicates the type of biomass predominant in each of these counties. We found qualitatively that each of the counties represented here has a major (>90%) category of waste biomass and no counties have even distributions of waste biomass types. Florida and California have a mixture of agricultural residues in the predominantly agricultural-waste biomass counties. However, the agricultural waste from counties in Nebraska/the Upper Midwest is almost solely represented by corn stover.

## Existing combustion facility retrofits

*Geographic distribution of promising retrofits and geologic storage*
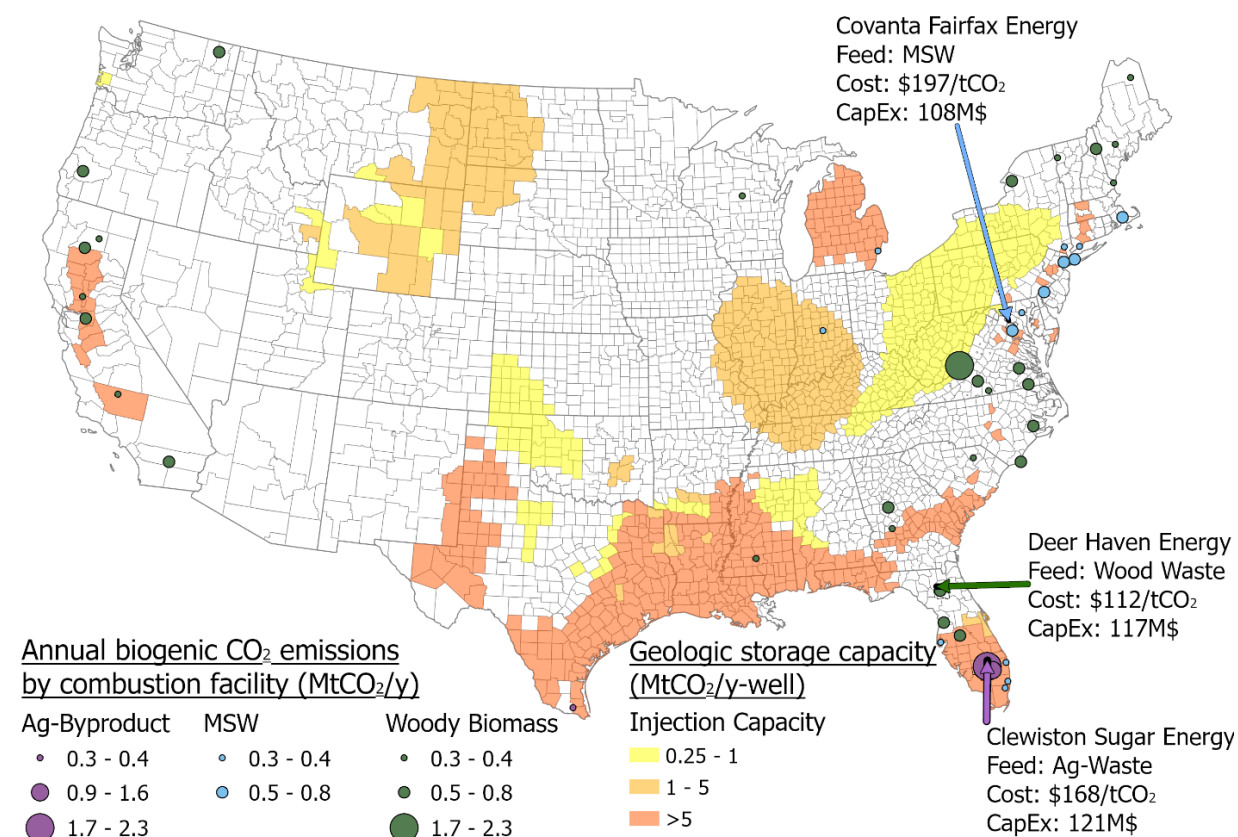


**Figure 2-9.** *Existing agricultural waste, MSW, and wood waste biomass-combustion facilities with annual emissions of at least 300,000 tCO_2 and regions of potential geologic storage capacity in the United States. Also indicated are the three facilities we analyzed in depth for retrofit costs.*

## Retrofit economics for theoretical facilities

In our analysis, we consider retrofits utilizing conventional absorber-stripper systems with two different liquid solvents: monoethanolamine (MEA), an older and more mature technology, and Cansolv, a more modern solvent system from Shell. Some evaluations of carbon-capture retrofits consider additional generation and even additional boilers to offset the generation losses associated with the energy required to operate the carbon-capture add-on, mostly used for solvent regeneration. In our analyses, we kept fuel supply constant unless otherwise stated and did not consider additional boilers, as we were focused on retrofit as a low-capital investment approach. Figure 2-10a shows the overall costs of $CO_2$ removal per $tCO_2$ in biomass combustion facilities with nameplate capacities ranging from 75–300 MW (operating at 85% capacity factor). The removal cost includes the annualized capital and additional operating expenses from the retrofit, as well as revenue losses from decreased power generation.
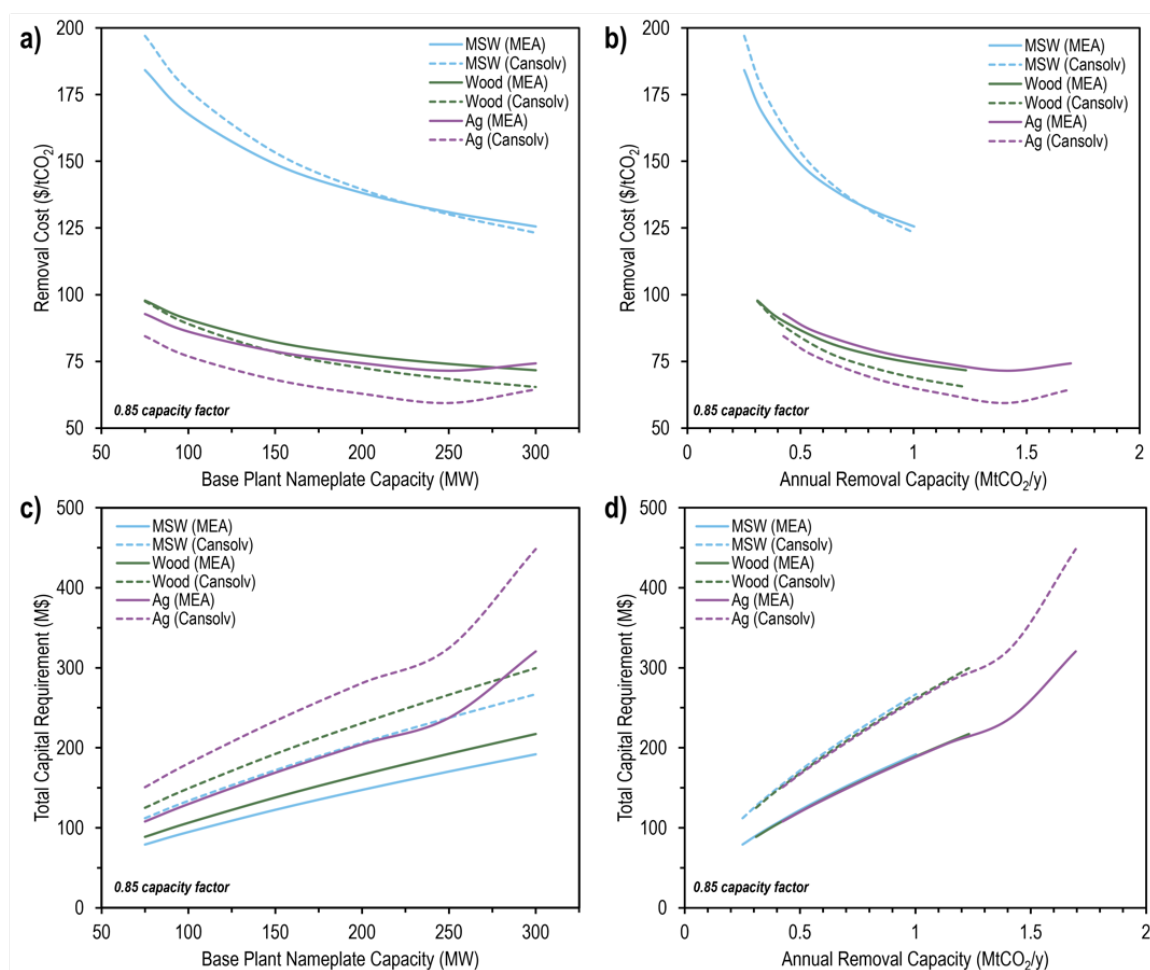


**Figure 2-10.** *Cost curves for carbon-capture retrofits using MEA and Cansolv solvents for theoretical biomass combustion facilities over a range of relevant nameplate generation capacities (a) and the associated annual removal potentials (b). Total effective capital requirements for these carbon-capture retrofits as a function of capacity (c) and removal potential (d).*

Between the two solvents we analyzed, MEA is generally the less capital-intensive technology (Fig. 2-10b). However, MEA is also less energy-efficient and significantly decreases the net generation of the combustion plants when fuel supply is held constant. In general, Cansolv typically results in lower overall removal costs due to lower generation losses and less lost revenue from the sale of electricity. The differences in capital requirements for the feedstocks we considered are related to heating values, moisture, and atomic composition of the feedstocks. In all cases, we did not consider feedstock cost because facilities are currently operating and are assumed to break even with established feedstock sourcing and contracts. Agriculture waste emits more $CO_2$ per unit energy generated due to differences in elemental composition and heating value and thus has greater capital requirements for a plant with the same capacity plant fed by wood or MSW. With more emissions to capture, a change in scaling factor is reached, leading to the bend in the cost curve for agriculture waste scenarios. However, overall, agriculture waste tends to have lower removal costs for a fixed nameplate capacity, as seen in Figure 2-10a, because of this greater $CO_2$ generation ratio.

For low-generation plants, the cost of removal for wood and agricultural waste combustion facilities are around or just below \$100/t$CO_2$. The overall costs are dependent on the selling price of electricity, which depends on existing contracts and the local grid power situation, as well as energy demands. In all cases shown, we assumed a selling price of \$0.06/kWh. As retrofitting relatively low-capacity facilities with carbon capture leads to a significant decrease in net generation, the selling price of electricity is a key consideration. As will be discussed further during the specific examples of retrofits, operation uptime (capacity factor) is another key factor that could be increased after retrofit to maintain the pre-retrofit net generation. However, this approach would require additional fuel and some facilities may be limited by their supply.

Figure 2-11 shows the loss in net generation and associated revenue losses for the different biomass feedstocks. Importantly, the fractional generation losses and revenue lost per $CO_2$ removed do not depend on facility scale in the capacity factor range considered here. As noted above, Cansolv is more energy-efficient and does not reduce generation as significantly as MEA. The revenue lost due to reduced generation primarily ranges from \$20–30/t$CO_2$, which is a smaller portion of the overall cost of removal in low-capacity facilities, which have high relative capital and operating expenses with less benefit from scale. The energy penalty of carbon capture for bioenergy problems can be partially offset with energy recovery in combined heat and power plants, but the reduction in losses is moderate (<5%) (*58*) and was not considered in any of our presented scenarios.
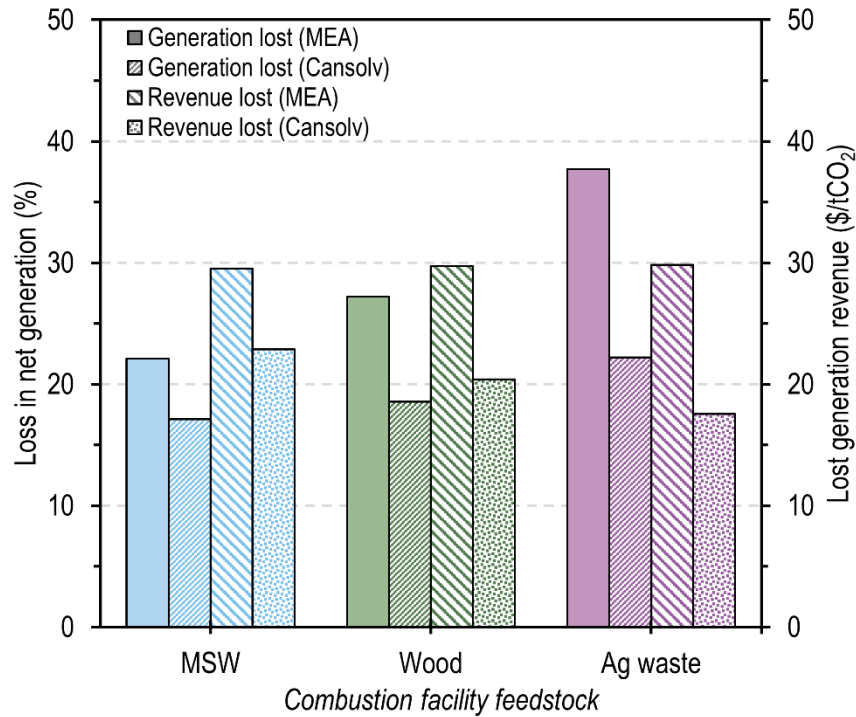
**Figure 2-11.** *Lost net generation of combustion facilities with carbon capture retrofit by feedstock as a percentage of pre-retrofit generation and the associated losses in electricity sale revenue per ton of $CO_2$ removed.*

## Carbon capture and storage (CCS) case studies for promising existing biomass combustion facilities

Finally, we performed several case studies for specific existing biomass combustion facilities, one for each of the three target feedstock types. We selected three facilities with relatively high nameplate capacities and geographic locations proximal to geologic storage regions: Deerhaven Renewable Energy Center (wood feedstock; Alachua County, FL), the Clewiston Sugar House (agricultural waste feedstock, Hendry County, FL), and Covanta Fairfax Energy (MSW feedstock, Fairfax County, VA). Table 2-6 shows facility nameplate and key retrofit results, including cost of removal, total capital cost, and loss in net generation. For each of these examples, MEA was the more cost-effective solvent system and had lower associated total capital costs. Notably, the agricultural biomass combustion facility, Clewiston Sugar House (Clewiston), has a low nameplate capacity and operates currently at a low capacity factor (32%). For each facility, we considered additional retrofit scenarios in which an increase in facility uptime (capacity factor) offset the loss in net generation from the addition of carbon capture. In these scenarios, the increased capacity factor was still lower than the standard, relatively high capacity factor used in generating the cost curves in Figure 2-10 (85%).

**Table 2-6.** Removal costs, capital requirements, and generation losses for each of the three facility retrofit case studies. The costs are for MEA retrofits, which was the solvent with a lower overall cost of removal in each case.

| | | | | Baseline fuel rate | | | Fuel ↑ to maintain generation | | |
|---|---|---|---|---|---|---|---|---|---|
| Fuel | Facility | Nameplate capacity (MW) | Capacity factor (%) | Retrofit removal cost ($/tCO$_2$) | Capital cost (M$) | Loss in generation (%) | Retrofit removal cost ($/tCO$_2$) | Capital cost (M$) | Loss in generation (%) |
| Wood | Deerhaven | 116 | 56 | 111.9 | 117 | 27 | 65.6 | 117 | 0 |
| Ag waste | Clewiston Sugar | 90 | 32 | 168.1 | 121 | 38 | 87.5 | 121 | 0 |
| MSW | Covanta Fairfax | 124 | 61 | 197.0 | 108 | 22 | 136.8 | 108 | 0 |

The most economical case among the three facilities is the wood combustion facility, Deerhaven Renewable Energy Center, with a post-retrofit removal cost of $112/tCO$_2$. Wood-fueled facilities have a moderate loss in generation with retrofit compared to agricultural waste and lack the biogenic fraction penalty associated with MSW combustion. Figure 2-11 shows a breakdown of each of the cases described and the associated costs. For increased capacity-factor scenarios, the facilities suffered no loss of generation revenue because we increased the overall generation to match the pre-retrofit scenario. However, these cases account for the cost of the additional fuel necessitated by increased capacity factor. The significant capital investments required are not affected by capacity factor, and greater plant utilization leads to only marginal operation cost increases but proportionally significant increases in generation and CO$_2$ produced and captured. As a result, the cost of CO$_2$ removal achieved by offsetting the retrofit with increased capacity is reduced significantly, typically by 25–40%.
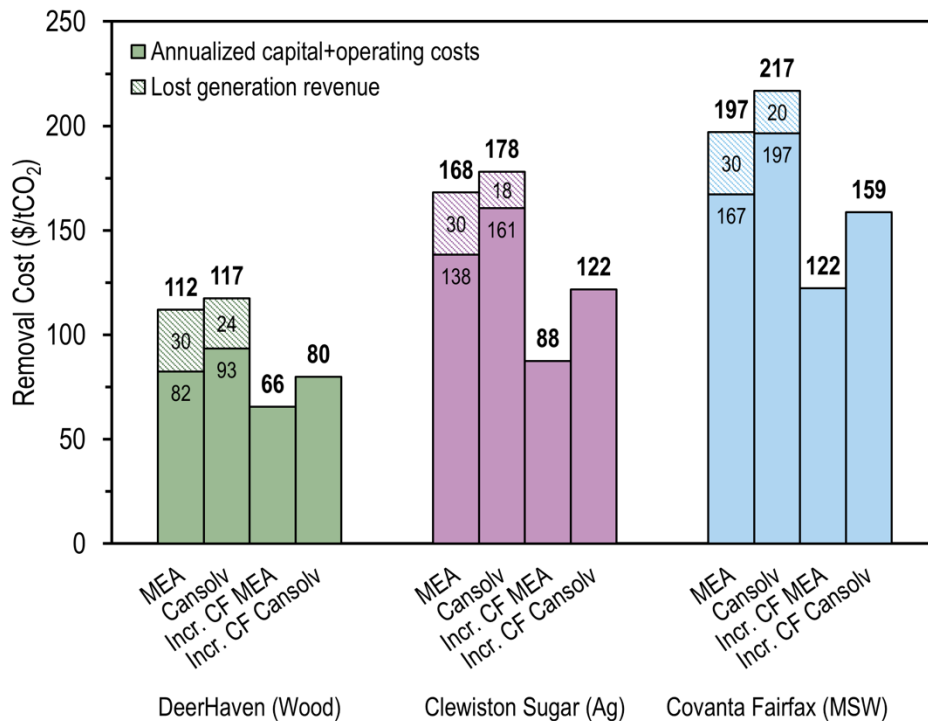
**Figure 2-12.** *Cost breakdown for four retrofit scenarios for each facility, using either MEA or Cansolv absorber systems and with or without increased operational uptime (capacity factor, CF) to offset generation losses from retrofit carbon-capture energy requirements.*

Very few agricultural biomass combustion facilities are in existence today, relative to the number of MSW and wood facilities. Waste-to-energy (MSW) facilities are numerous and located all around the country, especially near urban centers. These facilities are promising targets for carbon-capture retrofits for removal; however, the issue of biogenic fraction is very punitive when comparing overall removal costs with wood waste–fueled facilities. As a result, wood is the most promising feedstock in existing facilities amenable to retrofit for carbon capture. However, if MSW sourcing and pre-sorting could be organized such that MSW feedstocks were mostly or entirely biogenic, then MSW could be as promising if not more so than wood waste–fueled plants. As with co-firing, the existence of a cost associated with positive $CO_2$ emissions could further reduce this effect.

## CONCLUSIONS

We found that both biomass gasification to hydrogen and retrofit of existing biomass combustion facilities can provide millions of tons of negative $CO_2$ emissions at a price per $tCO_2$ significantly lower than today's DAC costs (estimated at $200/tCO_2$ (*5*)). New biomass gasification facilities could provide on the order of 200 $MtCO_2$/y if deployment and thus removal capacity met the quantities of available waste biomass. We found that the total system capacity of existing biomass combustion facility retrofits available at less than $200/tCO_2$ is around 12 $MtCO_2$/y.

Our analysis indicates that combustion facility retrofits for selected facilities are on the order of $100M in capital investment, whereas new construction of a gasification facility with similar $CO_2$ output (0.5

MtCO$_2$/y) is significantly more costly (on the order of $500M). We found that costs per tCO$_2$ follow the opposite trend: gasification to hydrogen provides negative CO$_2$ emissions at $25–50/tCO$_2$ at the smallest plant size, whereas retrofits of the most promising facilities at current operational capacity factors lead to negative CO$_2$ emissions at a cost of $100–200/tCO$_2$.

*We found that costs per tCO$_2$ for the gasification pathway can be minimized by selecting low-cost biomass, minimizing biomass transport costs (e.g., as represented by "regions of opportunity" in Figure 2-5), and maximizing plant size.* In particular, biogenic MSW—such as paper, paperboard, and construction debris—may provide seasonally consistent, low-cost biomass for gasification because processing (rather than landfilling) this biomass leads to avoidance of tipping fees. However, the price of this biomass was difficult to estimate—here we assumed a range of $0–40/dry ton—because we were not able to identify collection and sorting costs for these wastes.

Finally, *cost per tCO$_2$ for the retrofit pathway can be minimized by selecting the largest plants near geologic storage regions and increasing the capacity factor of the plant to counteract the energy penalty that results from adding carbon capture.* Currently, we find that wood waste is the most economical to process for retrofit plants, but pre-sorted biogenic MSW, if available, may be more economical and would enable more retrofit capacity across the United States.

# CHAPTER 3. SOIL CARBON

## SUMMARY

Agricultural soils have lost a vast amount of carbon to the atmosphere over human history (*6*). Even partially reversing this loss would remove a significant quantity of carbon from the atmosphere, thus slowing climate change (*7–9*). On the other hand, soils are dynamic natural systems that respond differently to management in different environments; hence efforts to increase soil carbon storage do not reliably yield net removal of $CO_2$ from the atmosphere in every agricultural or geographic context. In this section we will consider how these complexities influence the potential for soils to sequester carbon.

In addition to the broad criteria for carbon removal outlined above, we propose specific criteria that define effective soil-based carbon removal. For our analysis, we applied these criteria to five soil-management practices that have putative potential to achieve carbon removal: cover cropping, conversion of annual cropland to perennials, tillage reduction, organic amendments, and improved grazing management. We then mapped potential low-cost opportunities for soil-based climate mitigation at a county-level across the United States using the COMET Planner tool (*10*). Finally, we considered unresolved challenges related to soil carbon in the context of carbon credits.

## KEY FINDINGS

***Our major finding is that establishing verifiable and permanent $CO_2$ removal is challenging across all soil-based strategies; consequently soil-based carbon removal is a relatively high-risk method for offsetting $CO_2$ emissions. This risk does not obviate the need to manage soil in order to reduce greenhouse gas (GHG) emissions from agriculture, while potentially achieving $CO_2$ removal as a co-benefit.***

We developed first-order estimates for capacity and future costs of soil-based climate mitigation potential (defined as *removal plus avoided emissions*). We focused on cover cropping, which we ranked as having more clearly demonstrated climate mitigation potential than other practices based on the soil-specific quality criteria. Within the South-Central United States, where COMET Planner suggests that cover cropping might be most efficient at achieving climate mitigation, a maximum theoretical potential of 6 $MtCO_2e$/y of emissions might be avoided or removed from the atmosphere by this practice over a 10-year period. We estimate that costs of avoided emissions and removal associated with cover cropping in the South-Central United States are between $30 and $100 per $tCO_2e$ (9% discount rate) or $200/$tCO_2e$ (1% discount rate) assuming a 100-year target for permanence. Costs are sensitive to the risk of reversion to conventional management, which is not well constrained. ***We stress that these estimates combine avoided emissions and removal, which are not clearly differentiated in current soil carbon crediting protocols (6) and which should not be treated as equivalent quantities (7).*** The primary effect of best-practice management of agricultural soils is often reduced emissions, not net removal of carbon from the atmosphere (8, 9).

# INTRODUCTION

Practices that achieve net $CO_2$ removal via soil management rely on two fundamental strategies. Broadly, net-negative soil-management practices might first aim to increase the rate at which carbon is fixed from the atmosphere and introduced to soil (in this context, carbon fixation refers to the process by which $CO_2$ is converted and stored as energy or structure in living organisms). Additionally, net-negative soil-management practices might increase the persistence of soil carbon, allowing more to accumulate. Many climate-friendly agricultural practices also result in avoided emissions, which fall into three major categories:

1) **Avoided fossil fuel emissions**. For instance, replacing synthetic fertilizers with organic fertilizers may decrease carbon emissions associated with fertilizer manufacture and transport.

2) **Avoided emissions of GHGs with high global warming potential**. For instance, nitrogen applied in fertilizer can be transformed by soil microorganisms and emitted as nitrous oxide, which has a global warming potential 265–298 times that of $CO_2$. Reducing fertilizer use can reduce nitrous oxide emissions.

3) **Avoided soil carbon loss**. Some agricultural soils continue to lose organic carbon, and adopting improved management practices can slow, but not halt, continuing loss of this carbon (*59*).

Avoided emissions related to fossil fuel combustion and fertilizer application are easier to identify than carbon removal obtained by increasing soil carbon stocks and are also less susceptible to reversal (*60*). Consequently, avoided emissions are the primary pathway to mitigating the climate impacts of agriculture (*61*). However, given the objectives of this report, we emphasize strategies that target carbon removal in our analysis. Below we define six additional criteria for soil-based carbon removal.

## *Criterion 1:*
### The soil carbon stock must increase over time in excess of increases that would have occurred in the absence of improved management (*62–64*).

Given this definition, carbon removal depends on two reference values: 1) the initial carbon stock and 2) the carbon stock in the absence of the practice. Both numbers are required because soil carbon is not static. For instance, field experiments have shown that soil carbon stocks may decline even under improved management (*59, 62*). In these cases, relative improvements in soil carbon storage result in *avoided emissions* (Figure 3-1a). Alternatively, carbon stocks may increase even under conventional management (*65*); thus, *carbon removal* can occur with improved management but only after subtracting the soil storage increase that would have occurred if improved management had not occurred (Figure 3-1b). If the carbon stock declines under conventional management and increases under improved management, both avoided emissions and removal occur (Figure 3-1c). Importantly, this criterion depends on a robust estimate of the baseline change in carbon stocks that would have occurred in the absence of improved management.
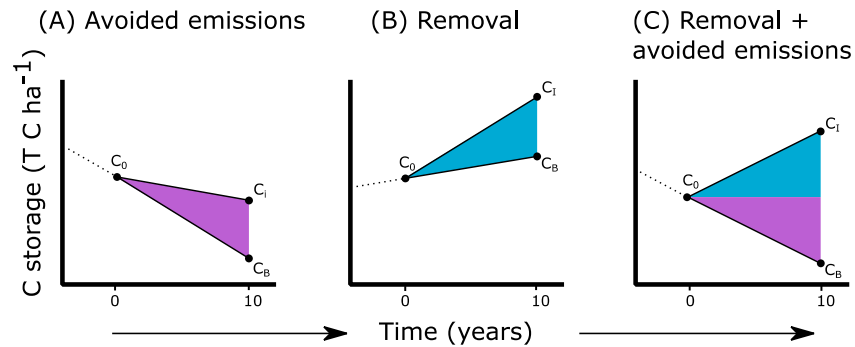
**Figure 3-1.** *Partitioning CO₂ removal. Diagrams show differences in carbon storage between an improved scenario (CI) and a baseline scenario (CB). Both scenarios begin at starting point C0. In panel (a) carbon is lost in both the improved and baseline scenarios and so only avoided emissions occur; in panel (b) carbon accrues in both scenarios and removal occurs; in panel (c) carbon accrues in the improved scenario and is lost in the baseline scenario, and so a mix of avoided emissions and removal occurs.*

## Criterion 2:

### The soil carbon stock must be evaluated over the full soil depth affected by management.

Soil carbon is often quantified at shallow depths (<30 cm) for monitoring soil fertility. However, management effects can extend to greater depths and can be negative or positive; hence the entire depth affected by management must be considered when reporting carbon stocks for climate mitigation purposes. Ideally the depth sampled should include the entire rooting depth of the crop; in practice, sampling should extend at least 10–20 cm below the typical depth of plowing (i.e., to 50 cm), given that management consistently affects carbon stocks directly below the plow layer (*66*, *67*). In addition, management can alter soil density and hence change the vertical dimensions of soil, so soil carbon stocks should be accounted for on an equivalent mass basis (*68*) and, at a minimum, soil bulk density must be quantified to identify possible sampling artifacts related to changes in density.

## Criterion 3:

### Increases in carbon storage must exceed associated emissions of other GHGs (methane, nitrous oxide, and fossil $CO_2$) on a $CO_2$-equivalent basis.

Nitrous oxide and methane are generated naturally in soils as microbial byproducts. In agricultural soils, nitrous oxide emissions are broadly linked to nitrogen inputs in fertilizer and manure. Most well-drained soils are methane sinks, but flooding soils for rice production produces methane, and at an agricultural-system level methane emissions are driven by livestock (enteric emissions and emissions from manure) (*69*). Nitrous oxide and methane are powerful GHGs; hence, increases in their emissions have the potential to offset or eliminate climate benefits from increased carbon storage (*70*). Conversely, reductions in emissions of these gasses, while beneficial, are avoided emissions rather than removal.

## Criterion 4:

### Practices must be evaluated at the scale of the regional economy to avoid leakage.

Management practices that reduce crop yields or displace agricultural production may drive agricultural intensification and land-use change elsewhere, compensating for lost production. Indirect land-use change may cause soil carbon losses that constitute leakage of emissions from the site where improved

51

management is applied. Furthermore, direct transfers of carbon from one location to another (e.g., in organic amendments) do not typically represent carbon sequestration; rather, they represent carbon redistribution (*64*, *71*), and hence can be considered a form of leakage.

### Criterion 5:

#### Management practices must be additional.

Actions achieve atmospheric carbon removal only if they sequester atmospheric carbon in addition to any sequestration that would have occurred in the absence of the action (the outcome must be additional). Consequently, payment for a practice that was already occurring for a significant timespan or would have occurred otherwise should not be used to claim climate benefits from that practice (*72*). In this chapter, we evaluate additionality by assessing current adoption rates for each agricultural practice, under the assumption that practices that are not already widely adopted are less likely to occur in absence of a payment.

### Criterion 6:

#### Increases in carbon storage must be maintained.

Soils are open biological systems that constantly return $CO_2$ to the atmosphere. While a fraction of soil carbon persists over decades to centuries, most of the carbon that enters soil exits it rapidly as $CO_2$ (*73*). All soil carbon is susceptible to decomposition and loss (*74*), and future increases in temperature are expected to increase the release of carbon from soil (*75*). Soil should not be viewed as a permanent sink for $CO_2$.

***Because soils are open systems (Box 3-1), exchanging a quantity of fossil fuel emissions for an equivalent quantity of soil carbon will have the net effect of transferring carbon from a highly persistent geologic reservoir to a reservoir where it is constantly emitted as $CO_2$. This fact does not negate the climate benefits of managing soils to increase carbon storage; however, it clearly indicates that fossil emissions and soil-based $CO_2$ removal should not be traded on an equivalent basis.***

# Box 3-1

## SOILS AS OPEN SYSTEMS

Soils store or lose carbon based on the balance of carbon inputs and carbon outputs. Consequently, any change in management that improves this balance can only yield increases in soil carbon temporarily; ultimately, the soil will approach a new carbon equilibrium when inputs and outputs balance. Despite this complexity, increases in soil carbon under different management regimes are typically reported in terms of either tons of carbon or tons of $CO_2$ equivalent (t$CO_2$e) per hectare per year, which implies that soil carbon stocks increase linearly. This convention is useful because soil carbon is often compared across experiments of different duration; however, in reality carbon storage rates are time dependent, with the fastest rates occurring earliest after a change in management (Figure 2). Any linear rates from the scientific literature reported in this chapter should be understood to represent the rate over the <10-year period typical of most agricultural experiments ("short term" effects), and rates over longer timescales are likely lower.

**Figure 3-2. Change in carbon storage versus average rate.** *Average rates of C storage in units of tCO₂e/ha/y mask the fact that soils are open systems and cannot gain carbon indefinitely. Typically, the rate at which soils accrue carbon after a change in management declines over time. Most agricultural field trials are <10 years in duration, hence long-term trajectories are unclear. Conceptual illustration adapted from Chenu et al., 2019.*
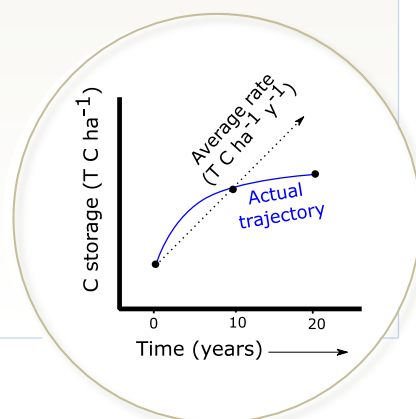
Keeping in mind that soil is not a permanent sink for $CO_2$, below we evaluate the relative persistence of carbon storage achieved by different soil-based climate mitigation strategies. We attempt to distinguish "fast cycling" soil carbon (<10-year residence time) from slower cycling carbon that turns over at multi-decadal timescales. While some soil carbon persists for centuries to millennia, there is currently no robust scientific basis for predicting how much newly added carbon achieves longer residence times; hence our assessments of relative persistence remain highly qualitative.

## Overview:

Considered together, these six criteria yield an overarching principle that is essential to evaluating carbon sequestration in soils: ***increases in soil carbon stocks do not necessarily yield net removal of GHGs from the atmosphere***. Baseline effects, GHG emissions, leakage, lack of additionality, and lack of permanence can all cancel the apparent climate benefits associated with a measurable increase in soil carbon relative to "business as usual." Consequently, demonstrating atmospheric carbon removal in soils is not simply a carbon quantification problem; ***rather, it also requires an integrated analysis that extends beyond proximate soil measurement***.

# ANALYSIS

## Comparison of soil-management strategies

Below we evaluate common soil-management strategies that have putative carbon removal benefits using the criteria above. We emphasize interventions that rely on existing technologies and practices, excluding those that are in early stages of development. We exclude biochar amendments; farms are a potential market for biochar that might offset production costs, but ultimately we see biochar as more appropriately grouped with bioenergy-based carbon removal strategies, given the upstream industrial infrastructure required (*76*). This leaves five broad categories of intervention: 1) cover cropping, 2) converting annual crops to perennial vegetation, 3) tillage reduction, 4) organic amendments, and 5) grazing management. Importantly, these five strategies all have primary environmental benefits independent of climate mitigation (e.g., erosion control and nutrient management). Climate mitigation is a potential secondary function of these practices.

### *Cover cropping*

Cover crops are planted on agricultural land that might otherwise be left bare, and cover crop biomass is typically returned to the soil rather than harvested and exported. Cover cropping is promoted as a strategy for increasing soil fertility and resilience because cover crops can improve the soil's physical characteristics, erosion resistance, and nutrient storage capacity. Planting cover crops has the potential to yield net $CO_2$ removal because cover crops fix carbon during fallow periods, increasing the rate at which atmospheric carbon is transformed into soil organic matter, a fraction of which may persist in soil over multiple years. In the context of U.S. agriculture, a primary opportunity for growing cover crops is during cooler winter months when cash crop production is less viable. Both nitrogen-fixing legumes (e.g., vetch, clover) and non-fixing crops (e.g., cereal rye) are grown as cover crops. Related management techniques include intercropping (maintaining cover in proximity with cash crops).

### Criteria 1-2: Evidence for carbon increase

Global meta analyses indicate that, on average, fields with cover crops have higher soil carbon stocks than fields without cover crops, although this effect is limited to the uppermost 30 cm of soil (mean short-term effect: 0.56 tC/ha/y) (*77*). This global average masks considerable local variation: at individual sites, carbon stocks may decrease when the whole soil profile is considered (*78*). Rigorous data syntheses that fully account for depth and employ equivalent soil-mass accounting methods are lacking for cover crops, which leaves the magnitude of soil-carbon change after cover cropping uncertain. In addition, the global average value cited above compares cover-cropped systems to conventional systems at a single point in time and hence combines avoided and negative emissions (Figure 3-1).

### Criterion 3: Greenhouse gases (GHGs)

On average, the overall capacity of cover cropping to improve the soil GHG balance appears to be positive (*79*, *80*); however, the effect varies. On average, leguminous (nitrogen-fixing) cover crops (e.g., vetch) appear to increase nitrous oxide emissions, whereas non-leguminous cover crops (e.g., rye) appear to reduce nitrous oxide emissions (*81*, *82*). GHG accounting approaches that incorporate indirect nitrous oxide emissions resulting from nitrogen in agricultural runoff indicate that cover crops have a

more universally positive effect on the GHG budget (*79*). On the other hand, over multi-decadal timescales, biogeochemical modeling suggests that accumulation of nitrogen and resulting nitrous oxide emissions under leguminous cover crops can turn cover crops into a net source of GHGs (*83*). These debates indicate that nitrous oxide emissions remain a critical uncertainty when evaluating cover cropping, even if the existing balance of information appears to show average reductions in emissions.

### Criterion 4: External inputs and leakage

Cover cropping does not depend on external biomass inputs, and we rate the economic leakage potential of winter cover cropping as moderate. Cover crops have, on average, neutral to slightly positive effects on crop yields (*84–86*); however, cover crops may compete with marketable crops or grazing opportunities (*84*), implying some potential for leakage. Competition with cash crops is likely to be higher in warmer climates, where cover crops might compete with opportunities to grow multiple crops in an annual cycle (e.g., in irrigated croplands in California or in the tropics). Some have argued that cover-crop seed production might displace food production (*87*), although this conclusion assumes—perhaps simplistically—that seed production would be spread evenly across land currently used to grow commodity crops and so should not be taken at face value.

### Criterion 5: Additionality

The additionality of cover cropping is geographically variable across the United States, but current adoption rates are low (less than 10%) in the Midwestern United States (*86*), which suggests that cover cropping may be relatively additional. County-level statistics from the 2017 USDA Census of Agriculture confirm that current cover-cropping adoption rates are relatively low (Figure 3-3). This low adoption rate suggests there are significant economic and cultural barriers to cover cropping, which is corroborated by local-scale economic surveys (*88*).

### Criterion 6: Relative persistence

Cover crops introduce plant residues to the upper part of the soil column. A fraction of the carbon may become associated with minerals, potentially yielding more persistent soil organic matter; however, most plant residue carbon is readily decomposed (*89*). This suggests that increases in the soil carbon stock due to cover cropping would be maintained most effectively if the practice is maintained year after year.

### Overall assessment:

While significant uncertainty surrounds the magnitude of soil carbon increases obtained through cover cropping, the overall average effect appears positive at the whole soil-profile scale; risks of increased GHG emissions, non-additionality, and leakage are real but moderate relative to other soil-based carbon removal practices. The simplicity of this practice—increased carbon fixation, yielding more soil carbon *in*

*situ*—is attractive; hence we will consider cover cropping as the example case in our expanded analysis in the Results section below.

## (A) Cover cropped

## (B) CRP grassland

## (C) No-till

## (D) Reduced till

Fraction of cultivated area

0    0.2    0.4    0.6    0.8    1
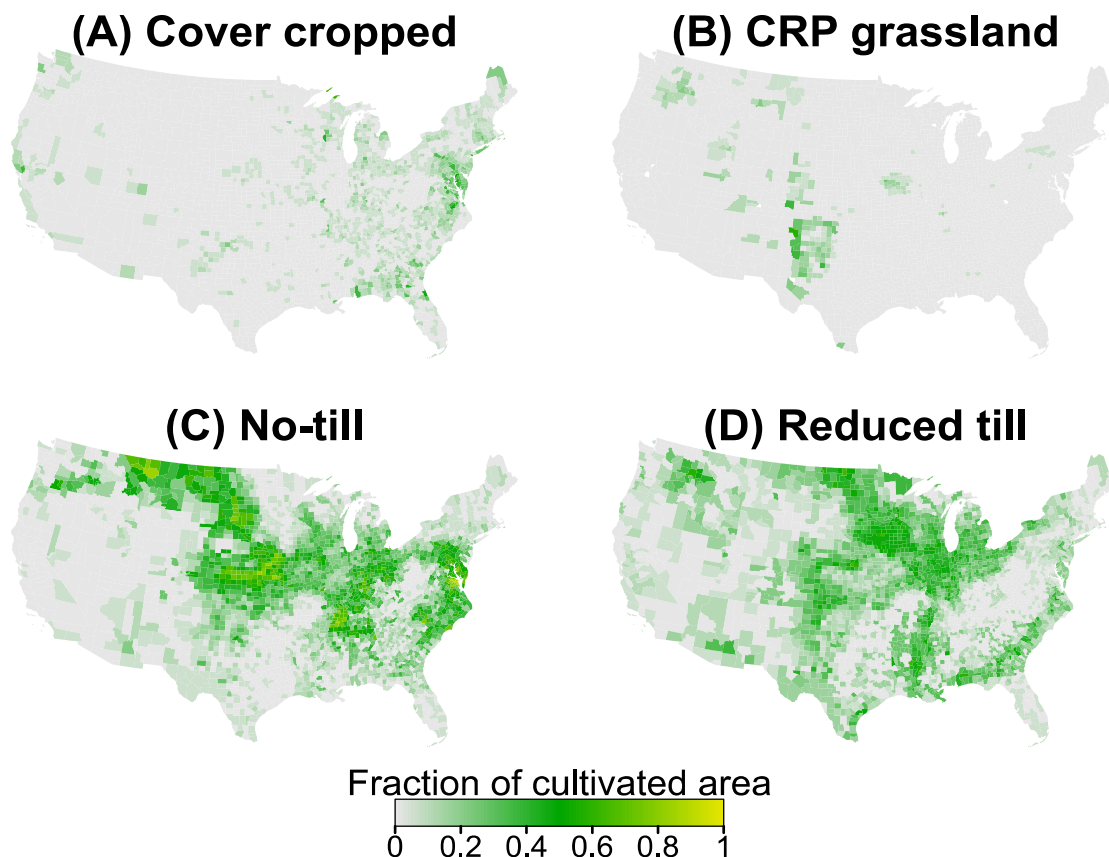
**Figure 3-3. Adoption rates for select practices.** *Maps show data for (a) cover cropping, (b) Conservation Reserve Program (CRP) grass plantings, and (c) no-till and (d) reduced-till farming. Colors show adoption as a fraction of total cultivated area from the USDA 2017 Census of Agriculture, with green representing 0%, yellow 50%, and red 100% adoption.*

## *Converting annual cropland to perennial grass cover*

Converting annual crops to perennial cover can yield significant increases in soil carbon due to an increase in year-round productivity, as well as an increase in root carbon and a decrease in disturbance from tillage (*90*). In this section we focus specifically on conversion of annual cropland to grassland or perennial grass bioenergy crops since conversion to forest is dealt with in a separate chapter.

### Criteria 1-2: Evidence for carbon increase

Meta-analyses suggests that converting farmland to grassland can yield 0.87 tC/ha/y (global short-term average) or 0.75 tC/ha/y (temperate latitude short-term average) relative to land maintained under cultivation (*91, 92*). In the United States, the USDA Conservation Reserve Program (CRP) covers payments to farmers for retiring marginal farmland and allowing it to return to perennial grass cover, primarily as an erosion control measure. CRP land yields short-term average relative soil organic carbon-

stock increases of 0.13 tC/ha/y in the Southern Great Plains (evaluated at 0–30 cm) (*93*). Perennial grasses invest more in root biomass than annual crops, yielding detectable changes in soil carbon well below the upper 30 cm of soil in perennial bioenergy systems (*94*, *95*). On the other hand, national-scale analyses of soil carbon-stock inventories suggest that deep carbon stocks are lower under CRP, offsetting gains at the surface (*96*)—although, whether this effect is due to land conversion or reflects underlying variation in soil properties that lead to land conversion is unclear. Average carbon-stock changes after conversion to perennials reported in meta-analyses are relative to paired controls and hence combine avoided and negative emissions. Net increases in soil carbon over time after conversion of cropland to prairie grasses have been documented at individual field sites (e.g., (*97*)).

### Criterion 3: Greenhouse gases (GHGs)
Lower nitrogen fertilizer inputs in perennial grassland versus conventional cropland has the potential to reduce nitrous oxide emissions (*98*). If fertilizer is applied to perennial grasslands, resulting nitrous oxide emissions may cancel carbon storage benefits (*99*).

### Criterion 4: External inputs and leakage
Replacing annual crops with perennial cover has potential to generate indirect emissions by driving land-use change elsewhere. In particular, conversion of arable land to bioenergy crops is responsible for net emissions related to indirect land-use change, which are very challenging to estimate (*100*). Focusing efforts on low productivity cropland ("marginal land") may mitigate this risk to an extent (*101*); furthermore, a significant fraction of U.S. cropland is already used to grow annual bioenergy crops, and replacing these crops with perennial bioenergy crops would minimize disruption to the biofuel market and might generate relatively less "leakage" (*102*).

### Criterion 5: Additionality
The CRP applies widely across U.S. croplands, but a significant CRP area has been converted to annual agriculture in the last several decades to meet demand for bioenergy products (*102*), and perennial cellulosic bioenergy crops are not yet widespread. These facts suggest room for growth of both conserved land (CRP) and perennial bioenergy, implying that these practices will be at least moderately additional. Currently, CRP grasslands are only regionally significant as a fraction of total cultivated land, being most common in regions of the South-Central United States that were affected by the Dust Bowl in the 1930s (Figure 3-3b).

### Criterion 6: Relative persistence
Perennial plants introduce carbon at depth via extensive roots, and carbon in deeper soil horizons (>30 cm) exchanges with the atmosphere slowly (*103*, *104*), which raises the possibility that soil carbon derived from perennial vegetation might be more persistent than carbon derived from shallow-rooted annual plants. On the other hand, while deep carbon cycles slowly on average, a component of deep soil carbon exchanges rapidly with the atmosphere (*105*), and roots can stimulate decomposition (*106*). The possibility that perennials yield persistent carbon is intriguing but requires more research.

## Overall assessment:

Conversion to perennials—as with cover cropping—yields increased carbon inputs to soil *in situ* and appears to have a robust effect. However, leakage risks are significant and probably require active management of the agricultural economy through government initiatives that fully account for the costs and benefits of indirect land-use change.

### *Tillage reduction*

Reducing or eliminating tillage is an effective erosion-control measure (*107*). Reducing tillage allows intact clumps of soil ("aggregates") to form. Aggregates are thought to have a role in carbon storage by protecting particulate organic matter from microbial attack, which raises the possibility that tillage reduction might increase soil carbon stocks.

### Criteria 1-2: Evidence for carbon increase

No fewer than 11 data syntheses have addressed the relationship between tillage and soil carbon storage within the past 5 years (*67*, *108–117*). Some studies have concluded that tillage has no statistically detectable effect on overall soil carbon storage (*108*, *109*, *111*, *113*), while others have identified positive effects (*110*, *116*, *117*) or have indicated that tillage



**Figure 3-4. Tillage effects on soil carbon.** *Data are from six synthesis papers that reported effect sizes as a function of depth.*

effects depend on climate and soil type (*67*, *110*, *116*). Considered at the whole soil profile scale, tillage reduction tends to increase carbon concentrations at the surface and reduce them at depth (Figure 3-4). This effect means that carbon storage in response to tillage reduction will be overestimated if it is assessed via shallow soil sampling (further details: https://carbonplan.org/research/soil-depth-sampling). The vast majority of data available on tillage effects document relative increases using paired studies that combine avoided emissions and carbon removal, although one recent meta-analysis (*116*) evaluated time-series measurements and found a positive effect of tillage reduction on soil carbon stocks over time.

### Criterion 3: Greenhouse gases (GHGs)

Implementing no-till or reduced-till farming appears to reduce nitrous oxide and methane emissions on average, although this response varies considerably (*118*, *119*). Other data syntheses have shown that increased nitrous oxide emissions following conversion to no-till largely cancel modest increases in soil carbon stocks (*80*).

### Criterion 4: External inputs and leakage

One meta-analysis considered the amount of crop residue left behind in no-till experiments and concluded that it is a major driver of apparent soil carbon gains (*120*). Crop residues that are removed under conventional tillage may be diverted to other fates (e.g., silage), and carbon in this biomass is not

necessarily immediately emitted as $CO_2$. Consequently, off-farm transfers of carbon should be considered when evaluating carbon sequestration from tillage reduction.

### Criterion 5: Additionality

In the United States, the additionality of tillage reduction is likely relatively low because no-till and reduced-till farming have been widely adopted (*121*). This finding is illustrated in Figure 3-3, which shows adoption rates near or above 50% across the major agricultural regions of the Central United States (Figure 3 C, D).

### Criterion 6: Relative persistence

Assuming that tillage reduction increases soil carbon stocks by promoting aggregation, much of the soil carbon is likely stored in aggregated-occluded particulates, which might be susceptible to loss if tillage resumed. Occasional tillage (i.e., every 5 years) does not have obvious negative impacts on soil carbon stocks (*122*).

### Overall assessment:

Carbon sequestration benefits of tillage reduction are highly controversial (*123*). Losses of carbon at depth are most common in cool temperate regions, which suggests that efforts to sequester carbon via tillage reduction could be targeted at warmer climates (*67*, *116*). We nonetheless concur with the authors of one recent meta-analysis (*67*), who argue that, given the uncertainties, any climate mitigation from tillage reduction should be seen as a welcome co-benefit rather than a primary function of this suite of practices.

### *Organic amendments*

Here we consider traditional organic amendments (e.g., manure, bio-solids, crop residues, compost), excluding biochar (see justification at the beginning of the Analysis section). Organic amendments divert waste biomass to soil, where it acts as slow-release fertilizer and improves soil structure and nutrient holding capacity. Applying organic amendments to soil locally increases soil carbon stocks (*124*), but the overall capacity of amendments to achieve net atmospheric carbon removal is less clear because the primary effect of amendments is local redistribution of carbon (*64*).

### Criteria 1-2: Evidence for carbon increase

Adding organic matter to soil is highly effective at increasing soil carbon stocks across the whole soil profile (*78*), and soil carbon increases due to amendments can be sustained over many decades (*124*). Criteria 1-2 are thus easily satisfied by organic amendments.

### Criterion 3: Greenhouse gases (GHGs)

Emissions of nitrous oxide per unit nitrogen are often lower for organic amendments than for synthetic fertilizers, which suggests that substituting organic fertilizer for synthetic fertilizer may reduce nitrous oxide emissions (*125*). The type of organic amendment strongly influences nitrous oxide emissions: amendments with a low carbon to nitrogen ratio tend to emit more nitrous oxide, and emissions also depend on climate and soil texture (*125*). In addition to nitrous oxide, organic amendments have the

potential to produce methane; however, diverting organic waste (e.g., manure) from anaerobic lagoons or landfills can have the net effect of reducing methane emissions (*126*).

### Criterion 4: External inputs and leakage

The carbon contained in organic amendments must originate from somewhere. ***Adding externally sourced carbon to soil does not necessarily constitute net carbon removal from the atmosphere given that biomass added to one place is biomass subtracted from another***. Thus, any potential for net removal depends on the alternative fates to which the biomass would have been subjected had it not been added to soil (*127*), which greatly complicates assessment of carbon removal associated with organic amendments. In theory, modeling the alternative fates of biomass and arriving at a full accounting of climate benefits is possible (*126*); however, in practice net removal of atmospheric carbon due to organic amendments is extremely challenging to verify with direct measurements.

### Criterion 5: Additionality

Organic amendments are applied to farmland where they are available; however, they represent a relatively minor fraction of total nitrogen applied in the agriculturally intensified regions of the Central United States. For instance, county-level data compiled by the U.S. Geological Survey indicate that, in sparsely cultivated parts of the Western and Eastern United States, the majority of nitrogen originating from manure and fertilizer comes from locally generated manure, but in the U.S. corn-belt only a minor fraction comes from this source (*128*) (Figure 3-5). The minor contribution of manure in agriculturally intensified regions reflects the imbalance between local manure supply and crop nitrogen demand. To the extent that meeting this shortfall requires importing manure, an additional unit of amendment applied in one place could entail subtraction from another (see Criterion 4 above).

### Criterion 6: Relative persistence

Studies from a long term experiment in Sweden used carbon isotopes to track amendment carbon in soil and suggest that a fraction of amendment carbon becomes associated with silt-sized mineral aggregates, which may persist at multi-decadal timescales (*129*, *130*).

### Overall assessment:

Organic amendments have major benefits for soil health. They can, under the right conditions, result in net carbon removal from the atmosphere (e.g., by increasing plant growth in unfertilized land (*131*)). However, establishing the net $CO_2$ removal achieved by a given unit of amendment is extremely difficult because it requires an accurate accounting of the alternative fate of the amendment biomass. Organic amendments



Manure-N, fraction of total N

0    0.2    0.4    0.6    0.8    1
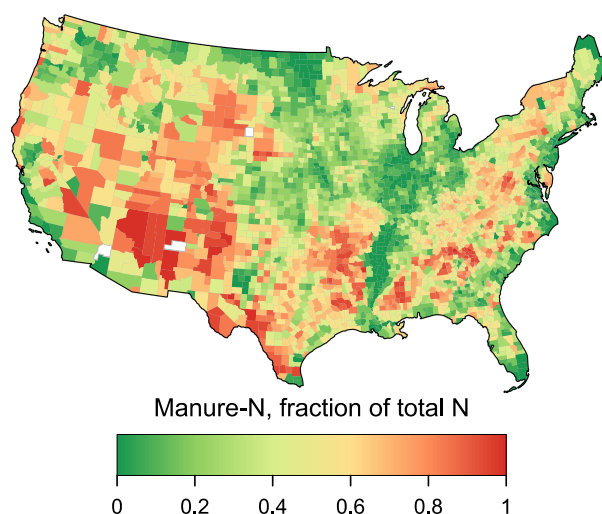
**Figure 3-5.** *Manure nitrogen in 2017 as a fraction of manure plus total synthetic nitrogen applied to farm and non-farm land. Data are from (69).*

should be supported on the basis of soil health and considerable emissions reductions benefits (e.g., avoided methane emissions (*126*)), rather than as a carbon removal strategy.

## *Grazing management*

This category encompasses a wide diversity of practices related to livestock grazing. These practices optimize pasture size, grazing duration, stocking rates, and resting intervals to maximize plant productivity. Increasing productivity—particularly carbon allocation to roots—has the potential to increase soil carbon stocks. Generalizing about the effects of grazing management on soil carbon storage is difficult due to the wide range of management variables involved.

### Criteria 1-2: Evidence for carbon increase

A recent global meta-analysis indicates that overall grazing intensity has significant effects on soil carbon stocks to at least a depth of 30 cm (*132*), although this study did not rigorously account for density changes, leaving the overall effect on soil carbon unclear. A multi-farm study conducted across the Southeastern United States found significantly higher soil carbon stocks under adaptive multi-paddock grazing (short-duration rotational grazing at high stocking densities) relative to conventional grazing. This study examined the whole soil profile using a relatively rigorous sampling design (*133*). Across the U.S. Great Plains, the *relative* effect of grazing management on soil carbon may be higher in lower rainfall environments, but the absolute effect is modest (*134*). The studies referred to above are based on paired comparisons, and thus cannot be used to evaluate net carbon increase over time. In addition, the definition of "business as usual" grazing intensity in the context of grazing management will likely have a strong influence on projected soil carbon gains.

### Criterion 3: Greenhouse gases (GHGs)

Ruminant livestock are a significant source of methane (via enteric fermentation and downstream manure) (*135*) and also drive soil nitrous oxide emissions through soil compaction and by concentrating and depositing mineral nitrogen in urine (*136*). Given that livestock mediate large fluxes of high-intensity GHGs, accounting for these gases is especially critical for identifying carbon sequestration in the context of grazing management. To the extent that improved grazing management techniques can reduce GHG emissions, these practices will benefit the climate system.

### Criterion 4: External inputs and leakage

With intensively managed pasture, livestock diets are often supplemented with imported biomass; on-pasture increases in carbon storage can be strongly sensitive to this external input (*137*).

### Criterion 5: Additionality

The varied nature of grazing management makes evaluating additionality difficult. Practices must be tailored to specific climate conditions; hence, identifying present adoption rates of improved grazing management practices is challenging.

### Criterion 6: Relative persistence

A multi-farm study conducted across the Southeastern United States found that adaptive multi-paddock grazing increases association between carbon and soil minerals, which suggests a relatively high degree

of persistence (*133*). However, relative carbon persistence cannot be directly estimated from this physical association.

Overall assessment:

Improved grazing management shows potential for increasing soil carbon stocks, but the overall net effect on the climate system is less clear due to the role of off-farm biomass transfers and emissions of methane and nitrous oxide. The climate benefits of various grazing management practices warrant further research, particularly in the domain of life cycle analysis. Importantly, avoiding conversion of rangelands to cropland has the potential to preserve soil carbon (*91*, *134*). Consequently, the biggest opportunities for managing soil carbon in rangelands may me more related to land use change than to the implementation of specific grazing practices in existing rangelands.

# Estimating capacity and costs

In sections 3 and 4.1, we developed criteria for evaluating the potential of soil-management practices to achieve carbon removal and then applied the criteria to five practices. This qualitative assessment revealed one practice that appears to have less uncertain climate benefits than the others: cover cropping. In this section, we evaluate the total climate mitigation potential (avoided emissions + removal) associated with cover cropping across the United States.

We estimated the climate mitigation potential of cover cropping using the COMET planner tool (*10*), which reports county-level data derived from the COMET biogeochemical model. Importantly, ***the COMET planner does not report accrual of carbon over time, only the difference between improved and baseline management scenarios; hence, it combines avoided emissions and carbon removal*** (Figure 3-1). In addition, COMET planner results represent 10-year average rates, which should not be extrapolated beyond this time window (Figure 3-2).

We used the COMET planner to obtain climate mitigation potential for Conservation Practice Standard 340 (cover cropping). We subtracted emissions reductions from nitrous oxide, methane, and fossil $CO_2$ emissions. Consequently, our estimates include carbon removal and avoided emissions associated with soil carbon-stock changes and are penalized for any increase in nitrous oxide, methane, or fossil $CO_2$ emissions without including reductions in emissions of these gasses.

To calculate the total climate mitigation potential in each county, we multiplied county-level COMET planner estimates ($tCO_2e$/ha/y) by the land area in each county that was cropped with two major commodity crops: corn and soy. We limited our analysis to land cultivated with corn or soy because they are summer crops; hence they are more obviously compatible with rotation of winter cover crops than other major commodity crops (e.g., dryland or winter wheat). We used the COMET implementation for non-legume cover cropping, given the uncertainties related to nitrous oxide emissions from legume cover crops. We derived available agricultural land area cropped with corn and soy from the USDA National Agricultural Statistics Service 2017 Census of Agriculture, subtracting out agricultural land that was reported as cover cropped in 2017.

Counties where cover cropping rates in 2017 exceeded 5% were excluded under the assumption that additionality is low once the practice exceeds this threshold (Figure 3-6). Notably, 5% is a relatively stringent threshold compared to the threshold used by existing soil carbon offset protocols, which set more relaxed thresholds for testing additionality (10–50%) (*72*). We applied a more stringent threshold than these protocols because our primary goal was to identify the geographic regions where additionality might be highest. Stringent accounting protocols might conceivably quantify additionality at a higher spatial resolution, rendering acreage in the counties that we excluded available.

We propagated uncertainty by developing two scenarios representing either a high biophysical potential, low-cost case (scenario 1) or a low biophysical potential, high-cost case (scenario 2). Uncertainty in the climate mitigation potential of cover cropping was fixed at +/-20% of the COMET planner estimates. We derived this relative uncertainty range from a recent meta-analysis (*77*) by dividing the 95% confidence interval for the global mean effect of cover cropping by the mean. We used this data-driven uncertainty estimate rather than the standard error estimates reported in the COMET planner because it reflects observed uncertainty in field trials, whereas the COMET planner uncertainty only reflects variation in the input parameters used to run the model.

# Cover crop rate > 5%
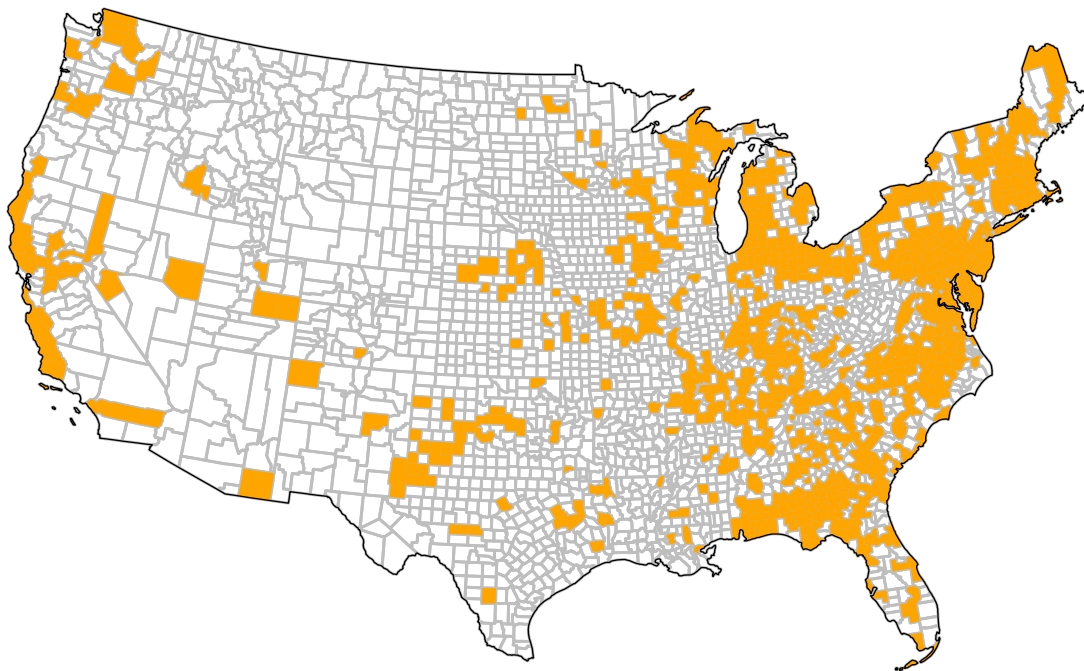


**Figure 3-6. Counties where the rate of cover cropping exceeds 5%.** *These counties were excluded from our capacity estimates because they exceeded the 5% additionality threshold.*

We obtained implementation costs for cover cropping using estimates from a study in Kansas (*84*), averaging upper and lower cost estimates across cover crop types (Table 3-1). While these data come from a single geographic locale, they are roughly consistent with estimates from a similar study in Iowa,

assuming that 100% of the cover crop biomass is returned to the soil and federal cost sharing incentives are not applied (*88*). We developed a range of monitoring cost estimates based on published case studies. Our lower cost estimate (scenario 1) was $32/ha and is based on mid-infrared proximal sensing (*61*, *138*), and our upper cost estimate (scenario 2) was $118/ha, based on composited samples analyzed using standard methods (*139*).

We modeled the time-integrated cost of implementing and monitoring soil carbon credits by assuming that carbon accrual is counted during an initial 10-year crediting period (in this case set by the 10-year boundary of the COMET planner simulations). During this period, monitoring occurs at 0, 5, and 10 years, and implementation costs are incurred annually. After this period, we assumed that soil carbon stocks reach equilibrium, crediting ceases, and payments for implementation cease. This hypothetical timeline assumes that cover crops eventually pay for themselves: farmers plant cover crops at a lower expense over time as they gain experience (e.g., (*88*)), and increased soil fertility and reduced fertilizer requirements from cover cropping accrue over time (e.g., (*140*)).

In each cost scenario, monitoring continues on a five-year basis for an addition 90 years to track 100-year permanence. During this 90-year period, reversion to conventional management may occur, voiding the credits and initiating another 10-year crediting period to replace them (a "pay-as-you-go" risk-management approach (*141*)). Our assumption that credits are immediately void following a management reversal is conservative, given that soil carbon stocks would not immediately revert to previous baseline conditions. We simulated reversal risk stochastically (inspired by (*142*)), assuming a 1% risk of reversion in the low-cost case (scenario 1) and a 5% risk in the high-cost case (scenario 2) (n = 10,000 simulations per scenario).

We chose risk parameters arbitrarily since we could not find information about the risks of management reversal. Alternative frameworks exist for addressing risk: in practice, soil carbon crediting protocols require buffer pools to ensure against reversal, although the size of the buffer pool is not well justified across protocols (*72*). The risk estimates we chose for the low-cost and high-cost scenarios increase costs by 4% and 18% relative to zero-risk versions of the scenarios, indicating that these are the buffer pool sizes corresponding to each scenario. ***More research is required to identify the risks associated with reversal of soil-based climate mitigation strategies and to develop accounting methods that directly account for reversal risk.***

In both the low-cost and high-cost scenarios, we time-discounted costs at a high rate (9%) representing returns on private capital. We also developed a third scenario using a lower social discount rate (1%) and a fourth scenario that was not time discounted, acknowledging that the choice of discount rate in the context of climate change is ethically fraught due to intergenerational transfer of costs (*143*). Other parameters were set to intermediate values in the third and fourth scenarios.

**Table 3-1.** Parameters used to develop cost estimates.

| Scenario | Biophysical effect modifier | Implementation cost ($/ha) | Monitoring cost ($/ha) | Risk of reversal | Discount rate |
|---|---|---|---|---|---|
| **1 Low cost** | +20% | $128 | $32 | 1% | 9% |
| **2 High cost** | -20% | $218 | $118 | 5% | 9% |
| **3 Low discount** | 0% | $173 | $75 | 3% | 1% |
| **4 No discount** | 0% | $173 | $75 | 3% | 0% |

## RESULTS

### Capacity and costs

The capacity of cover cropping to achieve climate mitigation (emissions reductions + removal) depends on two factors: 1) the amount of soil carbon that can be accrued relative to baseline management on a per-area basis and 2) the available agricultural land area. The importance of the second factor is evident
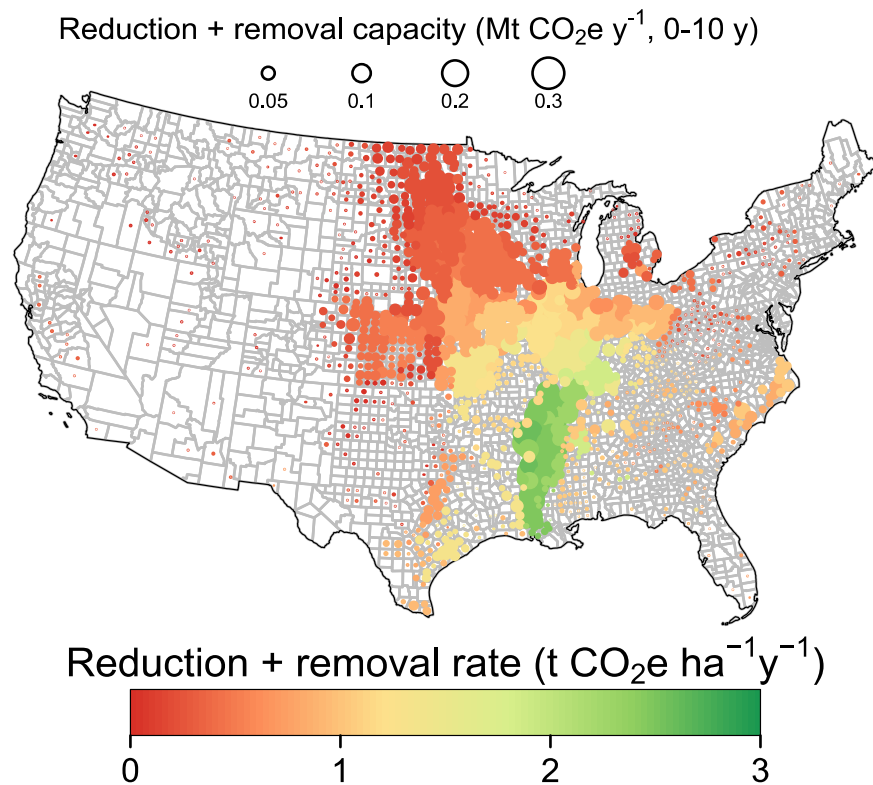


**Figure 3-7. Geographic distribution of emissions reduction + removal potential from cover cropping.** *The marker size indicates capacity in each county in millions of tons per year (MtCO2e y-1), while marker color indicates the per-area rate derived from COMET (tCO2e ha-1 yr-1). Counties where more than 5% of agricultural land is already cover cropped are excluded.*

in Figure 3-7, which shows that most capacity is in the heavily farmed Midwestern United States. Capacity is limited in colder and drier climates, where cover crop productivity is limited by lack of water or shorter growing seasons. This is supported by a meta-analysis that analyzed the response of soil carbon across multiple cover cropping experiments: generally soil carbon gains are greatest in warmer and wetter climates (*144*). Together with limited farm area, these climatic factors limit capacity in the Western and Northern United States. A third factor is the additionality constraint that we applied: a substantial number of counties in the Eastern United States were excluded because cover cropping rates were >5% in these locations (Figure 3-6). In principle, a fraction of this excluded area would be available if additionality could be constrained at a higher spatial resolution.

Without making any assumptions about implementation or monitoring costs, it is possible to consider the relative costs of climate mitigation achievable by cover cropping based on estimates derived from the COMET planner. Intuitively, many costs of managing soil carbon scale linearly with the land area managed; consequently, the cost in $/tCO$_2$e is likely lowest where the climate mitigation potential in tCO$_2$e/ha/y is highest. The opportunities with the highest per-area potential (and hence the lowest costs) lie in the lower Midwest and the lower Mississippi Valley (Figure 3-7). This result reflects the biophysical assumptions embedded in the COMET model: longer growing seasons in the humid Southern United States favor higher biomass accumulation and hence higher carbon inputs. Notably, the relatively warm climate in this region increases potential leakage risks (see Analysis section). ***There is likely an inherent tradeoff between the productivity of cover crops and leakage risk***: where cover crops are highly productive, the risk of displacing significant cash crop production is likely higher.

We derived cost estimates for the climate mitigation achievable by cover cropping across a range of hypothetical scenarios (Table 3-1). In Figure 3-8, we show the estimated cost in each county under each scenario versus the cumulative capacity, with counties ranked in increasing order of cost. The cost scenarios vary widely depending on the assumptions we applied, but all show a plateau in the initial section of the cost curve that corresponds to the least expensive capacity. ***This least expensive capacity is approximately 6 MtCO$_2$e/y for 10 years (Fig. 3-8). Critically, this figure represents a theoretical maximum; technical, economic, and social factors that are not reflected in our analysis will almost certainly limit the land area enrolled in soil-based climate mitigation efforts.***

The cost of the least expensive capacity is approximately $30/tCO$_2$e under low-cost assumptions and $100/tCO$_2$e under high-cost assumptions, in both cases assuming the costs are time discounted at the relatively high rate of 9%. If lower discount rates are applied, all costs well exceed $100/tCO$_2$e. This finding reflects the fact that carbon storage in ecosystems yields relatively short-term climate benefits coupled with long-term maintenance and risk-mitigation costs; hence ***present value costs of soil carbon storage are highly sensitive to reversal risk and discount rates***.

These projections rest on a specific set of cost assumptions, but in reality costs depend on many additional factors, including 1) the response of individual landowners to incentives, which depends on perceptions of risk, land tenure, and cultural factors, as well as techno-economic factors; 2) the type and duration of the financing scheme used to implement, monitor, and maintain soil carbon; and 3) the role of state and federal agricultural cost-sharing programs (e.g., EQIP). Cost sharing programs are particularly influential, given that current cost share payments can substantially offset implementation costs (*88*). Current cost-sharing schemes are not payments for carbon storage and, hence, may be an attractive option for offsetting costs without danger of double-counting carbon benefits. On the other hand, the existence of substantial federal cost-sharing incentives has significant potential to render practices non-additional.
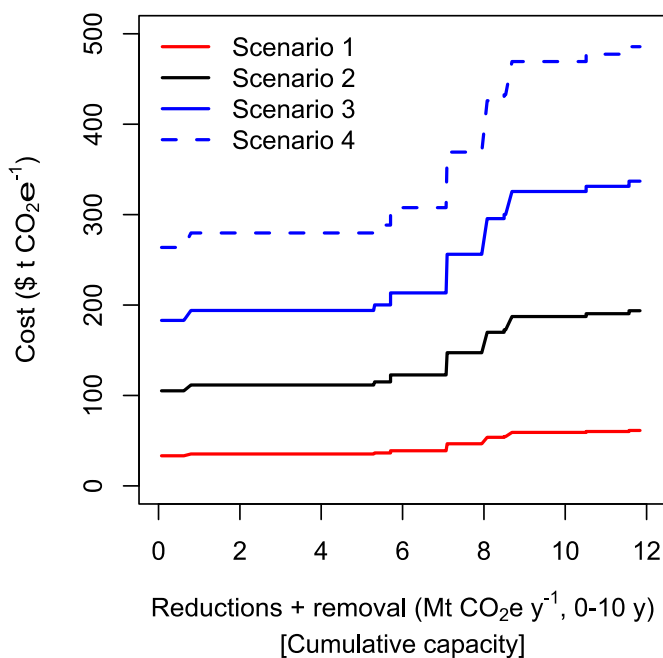


**Figure 3-8. Cumulative reduction + removal capacity for cover cropping versus estimated cost.** *Lines show the cumulative capacity derived by summing the capacity in each county, with counties ranked in increasing order by cost. Scenario 1 assumes higher per-area capacity, lower implementation costs, and lower risk, while Scenario 2 assumes the opposite (Table 1). Scenarios 3 and 4 assume 1% and 0% discount rates, respectively.*

# Technical risks and challenges

One major conclusion of this analysis is that soil-based carbon removal strategies are relatively high risk. Risk originates from several sources. One source of risk is reversal and impermanence, which has been discussed above (Introduction, Criterion 6). A second source of risk is systematic carbon accounting errors. Specifically, soil-management practices produce highly variable outcomes (*77*, *90*), and a large fraction of this variation cannot be explained by geographic factors or predicted with current biogeochemical models. This uncertainty means that **valuing carbon removal from soil-based strategies will depend on local measurements and locally implemented biogeochemical models**. Measurement artifacts and model under-calibration can potentially cause systematic errors in these estimates.

## *Measurement*
The gold standard for estimating soil carbon stock changes is on-the-ground measurements, which involve collecting soil samples from the field and quantifying soil bulk density and organic carbon (ideally via combustion analysis). However, even on-the-ground soil carbon measurement strategies have the potential to generate systematic errors. One primary example is the dependence of carbon

sequestration under no-till agriculture on sampling depth and reporting conventions (See Slessarev et al. 2021 at CarbonPlan: https://carbonplan.org/research/soil-depth-sampling). Shallow, fixed depth sampling can bias carbon-stock estimates. Best-practice ground-based sampling should include the entire soil depth affected by management (extending below the plow layer) and account for changes in soil density. Systematic landscape-level stratification is also crucial to ensure that samples are representative of the geographic area being characterized (*61*).

Similar issues affect remote sensing–based monitoring of soil carbon but to a greater degree. Remote sensing can estimate bare-soil carbon concentrations with low to moderate accuracy (*145*) but only at the soil surface and without accounting for changes in soil density or soil rock fraction. Changes in the vertical distribution of carbon, soil density, and soil rock fraction all significantly affect carbon-stock estimates, resulting in **high potential for remote sensing–based monitoring to produce systematically biased estimates**. These technologies may however have a role in informing ground-based estimates— for instance, in defining stratified field sampling strategies or verifying compliance with the target management regime.

### Biogeochemical models

Models provide a second avenue for systematic errors in soil carbon-stock change estimates. The soil biogeochemical models used for simulating carbon stocks in agricultural systems are based on theoretical assumptions that are essentially unchanged since the 1980s, which assume a linear system of operationally defined carbon pools that cycle at rates modified by climate and soil texture. **Conventional, linear model structures are unsupported by the last several decades of empirical biogeochemical research** (*146*, *147*). This issue has serious implications for forecasting carbon-stock changes. For instance, conventional models predict that carbon stocks increase linearly with the rate of carbon input, whereas in reality carbon stocks increase at a diminishing rate as the input rate increases (*148*). Conventional models also systematically underestimate the age of soil organic matter (*149*). This finding implies that, while average soil carbon residence times are long, the fraction of new carbon that can attain a long residence time is lower than conventional models suggest (*149*). In addition, conventional models—specifically the model employed in this analysis—have very low predictive power with respect to nitrous oxide emissions (*150*).

More sophisticated nonlinear models are still largely experimental and produce highly divergent outputs (*151*); hence, conventional biogeochemical models may be preferable given that they have been more widely validated. However, these models should generally be considered provisional, approximate tools (including the COMET outputs used here), and validation data remain insufficient to allow confidence in fine-scale predictions (*61*). **Soil carbon credits should not be valued based on model outputs alone**.

### Life cycle analysis

Other risks lie in improper accounting at a systems level. For instance, the fact that organic amendment transfers to soil do not yield equivalent carbon removal is not widely appreciated and can yield gross overestimates of their climate mitigation potential. Organic amendments are commonplace in agriculture and can be applied in concert with other practices. This issue raises the possibility that carbon in organic amendments might be misattributed as sequestered if amendments are applied in

concert with another practice. Monitoring protocols applied at a farm-level will thus need to account for the farm-level organic matter budget, regardless of whether amendments are being directly credited. This "systems level" perspective is essential to high-integrity monitoring of soil carbon.

## CONCLUSIONS

For soil carbon management to achieve carbon removal, a broad range of criteria must be satisfied: 1) soil carbon must accrue over time beyond any accrual that would have occurred naturally, 2) soils must be evaluated at the full depth affected by management, 3) the carbon storage increase must exceed cumulative $CO_2$-equivalent emissions of other GHGs, 4) externally sourced biomass and economic leakage must be accounted for, 5) practices must be additional, and 6) carbon gains must be maintained. ***Even if these criteria are all satisfied, carbon stored in soil is biologically cycled and hence intrinsically more vulnerable than carbon stored in geologic reservoirs.***

With these caveats in mind, the near-term theoretical maximum capacity for combined emissions reductions and (temporary) carbon removal from soils is possibly quite large, with the least expensive opportunities equaling 6 million $tCO_2e$ per year for a single practice (cover cropping) over a 10-year period. The predictions of the COMET planner tool indicate that geographic opportunities are more southerly within the United States. Efforts to exploit this potential should prioritize ground-based monitoring and should not rely entirely on models or remote-sensing measurements.

***As a concluding message, we emphasize that many soil-management practices that have limited or uncertain potential for net carbon removal have significant potential for achieving avoided emissions and yield significant benefits to soil fertility, water quality, and sustainability of the broader agricultural system. These benefits alone are compelling reasons to support improvements in agriculture outside the narrow bounds set by emissions markets.***

# CHAPTER 4. FORESTRY

## SUMMARY

Forestry is an often-cited intervention for avoiding carbon emissions and/or removing carbon from the atmosphere. Aside from the ecological benefits of preserving or restoring forests, forestry offsets appear attractive from a price standpoint, as they typically cost less than technological carbon removal solutions. Among natural climate solutions, forestry occupies central stage because of its large potential scale. Here, we investigate the physical feasibility of removing $CO_2$ from the atmosphere at a multi-million-ton-per-year scale through forestry, i.e., a scale that is meaningful for corporate first-movers in the carbon removal space. We also look at related challenges, such as transactional volume, baseline definition, additionality, leakage, and durability. This analysis was based on literature review and consultation with sectoral and topic experts.

## KEY FINDINGS

Managing forests for the purpose of carbon removal can have ecological and landscape co-benefits, making the pursuit of projects desirable for reasons beyond carbon. However, despite the large theoretical potential to sequester $CO_2$ in U.S. forests, several challenges stand in the way of achieving removals at the multi-million-ton-per-year scale through forestry. These challenges include the potentially very large number of landowners and transactions involved—a logistical challenge that may be mitigated in the future through technology. They also include difficulties in defining project baselines, establishing additionality, and quantifying credits appropriately, concerns over induced impacts elsewhere (leakage), and a relatively high risk of reversal—accounting challenges that will require significant effort from project practitioners, researchers, registries, and governments to overcome. Technology and improved project governance and accounting structures may be able to assist with some of these challenges but likely not with durability concerns.

Therefore, despite its apparent attractiveness, the forestry sector must address challenges related to definition of baselines, additionality, leakage, and durability (including risk of reversal due to fire) in order to be a sound backbone for corporate carbon removals at the needed scale. As such, forest carbon credits should be used judiciously, with individualized due diligence at the project level and more thorough safeguards and frameworks than are publicly available today.

As with soil carbon sequestration, the challenges of forest carbon projects do not obviate the need to protect and manage forests in order to maintain existing carbon stocks, sinks, ecosystems, and landscapes, while potentially achieving other co-benefits.

## INTRODUCTION

For this chapter, we investigated the feasibility of removing $CO_2$ from the atmosphere at a multi-million-ton-per-year scale through forestry, arguably the front-runner among natural solutions, due both to its potential scale and to its lower cost (*152*).

## TYPES OF FORESTRY PROJECTS AND OFFSETS

We examined the following categories of forestry projects and offsets to determine their potential for carbon removal:

1. **Avoided deforestation.** This intervention prevents forest clearing or destruction, avoiding a baseline whereby an existing carbon sink would be permanently degraded. Avoided deforestation typically qualifies as avoided emissions. An avoided deforestation project can also maintain a natural sink and hence an ongoing rate of carbon removal within its geographic boundaries. However, maintaining this sink does not result in new carbon removal from the atmosphere but rather prevents backsliding of existing, ongoing removal. In fact, the existing land sink is typically taken into account in global climate assessments and greenhouse gas (GHG) balances. Although avoiding deforestation is clearly desirable and essential for controlling climate change, quantifying carbon removal, ensuring its durability, attributing the net effect to a particular project, and ensuring that no harm is done elsewhere as a result, all involve considerable difficulties, which we examine below.

2. **Improved forest management (IFM).** In contrast to avoided deforestation projects, which prevent the complete loss of forest cover, IFM projects typically involve managed harvesting to achieve a deviation from an existing baseline in areas that are likely to remain as managed forests. The intervention commonly seeks to extend the life of trees or delay their harvest—thus avoiding the sink degradation—and also to increase the sequestered carbon in the area of interest by letting trees grow larger. Other approaches to IFM include thinning for overall forest health, reduced impact logging, and temporary increases in rotation length. In addition to an avoided emissions component (in the same manner as avoided deforestation projects above), IFM can also have a carbon removal component because the rate at which existing individual trees remove carbon from the atmosphere is increased as they are allowed to grow bigger. Subject to certain size and saturation limits, an individual tree will generally absorb more carbon from the atmosphere annually as it grows larger. It is important to note that this is true at the individual tree level but not the stand level. However, even if the rate of storage decreases in stands, they still continue to store carbon for decades. See (*153*). However, projects vary greatly in nature, quality, and duration. Baseline definition and determination of leakage (or lack thereof) can be challenging.

3. **Afforestation/reforestation.** This intervention, which restores forest or establishes new forest lands in areas where they do not currently exist, is typically the easiest to link to carbon removals, although indirect leakage effects and displacement need to be considered. Quantifying carbon removal, which in this case corresponds to increased sequestration, is easiest for this category than for other categories of forestry projects.

## NUMBER OF LANDOWNERS: A LOGISTICAL CHALLENGE INHERENT IN FORESTRY CARBON PROJECTS

Studies point to a large theoretical potential for increasing carbon sequestration in U.S. forests. For example., Domke et al. (*154*) estimate that 187.7 $MtCO_2$/y [±9.1] can be sequestered through tree planting in understocked timber lands at ~1 $tCO_2$/acre/y. This is on the low side because understocked timber lands are a special case. Fargione et al.(*155*) estimate a reforestation potential of 307 $MtCO_2$/y on ~155M acres (high uncertainty) at ~2 $tCO_2$/acre/y and another 74 $MtCO_2$/y on 40M acres of

pastureland. Cameron et al. (*152*) present a range of approximately 2–7 $tCO_2$/acre/y depending on the species and the year (sequestration rates increase as the trees grow). The particular rate is very dependent on species, climate soil type, and age.

It is worth noting that these numbers represent maximum potential and do not account for "leakage" as a result of market dynamics, for example when previous land uses are displaced to outside the project area. The issue of leakage is discussed further below.

Even if the total potential numbers for sequestration are large, the amount of $CO_2$ sequestered per acre in these estimates is modest, underscoring one of the big challenges with forestry projects: a large amount of land area must be recruited.
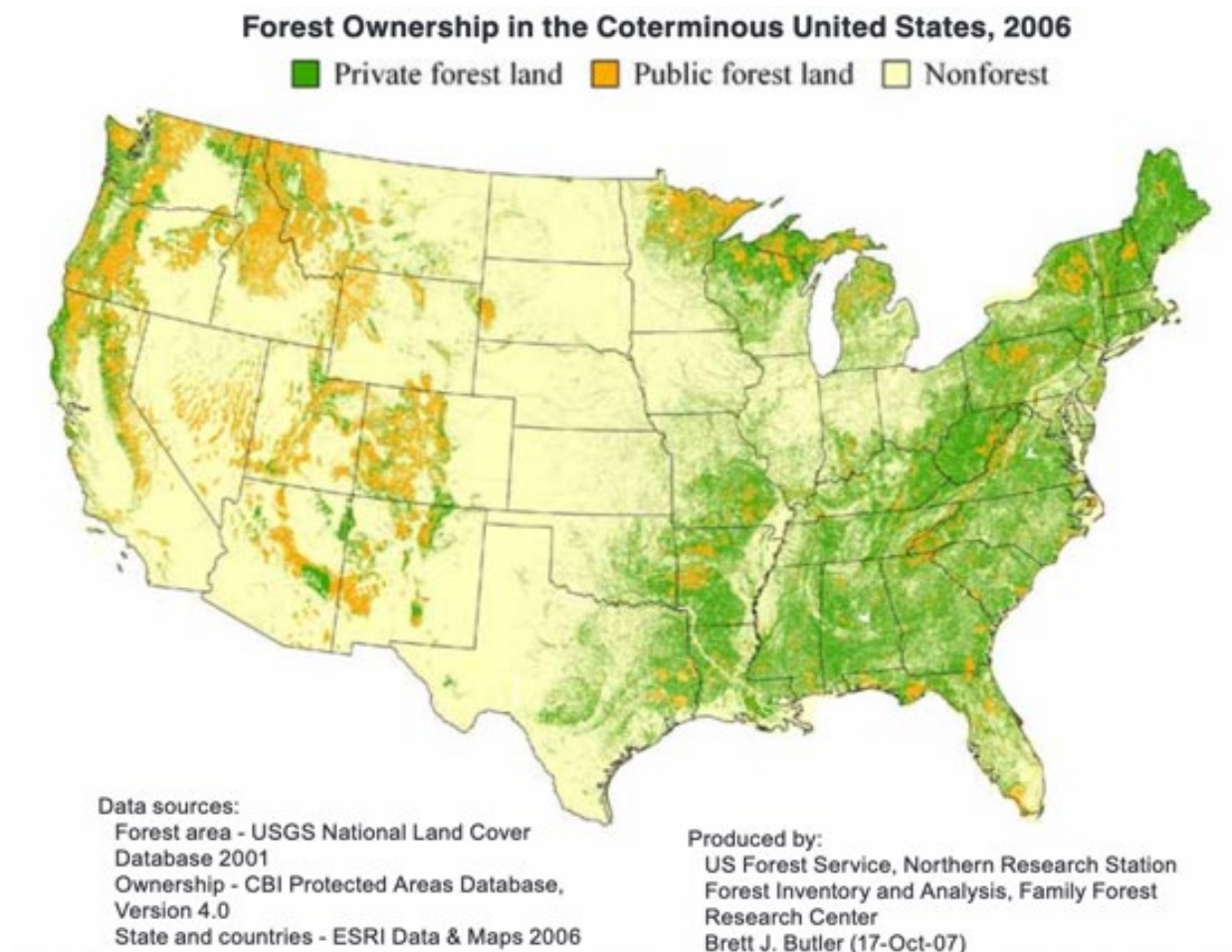


**Figure 4-1.** *Forest ownership in the coterminous United States according to the U.S. Forest Service. (156)*

**Table 4-1** – Acreage and Ownership of America's Family Forests.

| Size of forest holdings (acres) | Number of acres | Number of ownerships | Average number of ownerships needed within each holding size for 1 MtCO$_2$/y |
|:---:|:---:|:---:|:---:|
| 1–9 | 19,158,000 | 6,221,000 | 162,360 |
| 10–49 | 58,585,000 | 2,832,000 | 24,170 |
| 50–99 | 41,562,000 | 644,000 | 7747 |
| 100–999 | 97,667,000 | 508,000 | 2601 |
| 1000+ | 35,003,000 | 19,000 | 271 |
| Total | 251,975,000 | 10,224,000 | |

Figure 4-1 shows that private landowners hold the larger portion of forests in the coterminous United States, and Table 4-1 shows that the number of such owners is very large. As a sample calculation of the number of landowners that would be involved to achieve a total sequestration of 1 MtCO$_2$/y (assuming a moderately conservative sequestration rate of 2 tCO$_2$/acre/y), we assumed a distribution of forest holding size in our sample forestry carbon removal portfolio of 20% with a holding size of 1–9 acres, 30% with a holding size of 10–49 acres, 25% with a holding size of 50–99 acres, 20% with a holding size of 100–999 acres, and 5% with a holding size of over 1000 acres. This calculation yields a total of *42,194 landowners* needed to achieve a total sequestration of 1 MtCO$_2$/y.

For comparison, sequestration of 1 MtCO$_2$/y would require only 271 landowner participants if projects included only the largest class of landowners (i.e., portfolios only included holdings larger than 1000 acres), assuming equal-sized land holdings between the 19,000 owners with holdings this large. Similarly, 162,360 landowner participants would be required if only the smallest holding-size class (1-9 acres) were included. The last column of Forestry Table 1 shows the number of landowner participants required for each holding-size class if it were the only class involved in a 1 MtCO$_2$/y project.

In summary, any endeavor to sequester CO$_2$ at the million-ton scale using forestry in the United States would likely need to involve millions of acres. Unless a very high degree of the nation's largest landowners is achieved, this translates to hundreds to hundreds of thousands of landowners. This low density is one of the most fundamental challenges inherent in forestry projects, as the large land area required may be hard to recruit and manage, and the number of transactions required with owners of that land poses a logistical challenge. Technology may be able to facilitate these transactions in the coming years.

# CHALLENGES INVOLVED IN QUANTIFYING THE TRUE CARBON BENEFIT OF FORESTRY PROJECTS

Forestry projects typically face additional inherent challenges beyond the number of landowners involved. The biggest challenges are related to quantifying the true carbon benefit of projects. Among these challenges are 1) defining the *baseline* against which project benefits are measured, 2) determining the *additionality* of the project (or lack thereof), 3) determining whether the project has direct or indirect effects or is subject to market or other pressures that diminish its overall benefit (*leakage*), and 4) predicting the *durability* of the carbon removal.

A project's *baseline* is the reference case and refers to the quantification of carbon stocks in the project area before any action is taken, and determination of what would have happened to carbon within the project's geographic boundary in the absence of the offset program (see, e.g., relevant discussion in Climate Action Reserve: https://www.climateactionreserve.org/how/protocols/forest/baseline/). For example, for an improved forest management project (IFM), the baseline may involve harvesting a set percentage of trees of a certain size at a particular frequency. For an avoided deforestation project, the baseline may involve clear-cutting the entire area within the project boundary. *Additionality* is an issue that is linked to determining the baseline and refers to whether the action taken as a result of the project would have taken place anyway, even in the absence of the project. A project may have an accurate baseline, but still be non-additional. For example, claiming that the default fate of a particular area would have been commercial harvesting when neighboring land is going out of economic production may raise additionality concerns for a proposed IFM project. The same could be said of a proposed IFM project in an area that has been protected by a land conservation organization for years— the default fate is likely not harvesting and the project would not be additional to what would have happened in its absence. It is worth noting that the challenges in determining baselines and additionality may be scientific (e.g., accurately quantifying existing stocks or existing harvesting rates) and/or accounting-related (e.g., whether a particular offset protocol defaults to a particular stock density once an arbitrary geographic boundary is crossed or assumes a fixed rate of harvesting without taking into account local particulars).

*Leakage* refers to emissions that are shifted elsewhere as a result of the project, despite reductions within a project's boundary (*157*),(*158*) . In the forestry context, the carbon benefit of a particular project may be reduced or negated entirely if an area equivalent to that protected by the project is simply felled elsewhere (*159*), (*160*), (*161*), (*162*), (*163*). In the case of an avoided deforestation project, leakage may come in the form of another area being harvested instead; in an IFM project, leakage may occur when the timber that is now not commercially harvested as a result of the project is provided instead from land areas that are not within any offset project boundaries and devoid of protections.

*Durability* is an often-cited concern with forestry projects. Tree, being living organisms, are subject to stressors such as drought, fire, pests, and disease, which can affect their rate of carbon uptake or even completely kill them. As such, there is a distinct risk of reversal for many of these projects. This is sometimes dealt with through buffer pools that offset registries require, which are credit reserves set aside to make the system whole in the event of some project reversals. However, these may be too

small to cover the entire set of reversals, particularly in the event of a natural catastrophe like wildfire or drought-induced mortality. Climate change is also exacerbating these concerns by giving rise to more frequent and more severe conditions that can lead to reversal (see, e.g., discussion in: https://climatographer.substack.com/p/do-we-need-to-rethink-nature-based).

Defining baselines and determining leakage or lack thereof can be notoriously difficult in the context of forestry offsets. Thus, the validity of forestry offset projects has been called into question several times. Of note, a recent analysis by Carbon Plan concluded systematic over-crediting of forest offset projects (*164*). We believe that a conservative approach toward claiming carbon removal through forestry projects by corporate first movers in the carbon removal space merits a close, project-by-project look and discussion of several specific parameters with the project host. One possible temporary approach for minimizing complexity in baseline and leakage considerations would be to restrict forestry offsets to reforestation and afforestation projects on unambiguously degraded land with no competing uses and with accurate accounting of natural regeneration; however, land like this can be scarce since land that can support healthy forests is likely to have some competing use.

## THE ROLE OF TECHNOLOGY IN FORESTRY OFFSETS

We also examined technological approaches taken by some forestry project companies, specifically those that use remote sensing techniques such as radar, LiDAR, and satellite imagery to quantify forest carbon. We conclude that a combination of these techniques can indeed successfully quantify forest carbon and that, in fact, some techniques may be excessive (e.g., full-resolution 3-dimensional LiDAR). However, this capability alone cannot today overcome the inherent challenges that forestry projects face, as outlined previously. It is possible that, within a few years, accurate quantification of forest carbon stocks will be mainstream and standardized, and that transactional costs and logistics will be reduced to a degree that makes the involvement of very large numbers of landowners in forestry projects feasible. However, we expect the challenges related to baselines, additionality, leakage, and durability to remain, absent significant systemic reform.

## AVAILABILITY OF U.S. FORESTRY OFFSETS BY TYPE

We used U.C. Berkeley's recently published global voluntary offsets database to summarize the forestry offsets issued in the United States since 2009. We selected 2009 because it is the year the first, and still largest, afforestation/reforestation project (*Green Trees*, or ACR114) started and also roughly the span of a decade until the time of writing. Around 20% of issued credits are held in a buffer account to cover the risk of reversal.

**Table 4-2** – Forestry offsets in the United States since 2009, as of August 2021. 1 credit corresponds to 1 tCO$_2$e. Source: https://gspp.berkeley.edu/faculty-and-impact/centers/cepp/projects/berkeley-carbon-trading-project/offsets-database

| Type | Credits issued | Credits not retired (e.g., held in a buffer pool for reversals) | Number of projects with first credit issuance between 2009–2020 |
|---|---|---|---|
| **Afforestation/reforestation** | 5,178,289 | 2,925,527 | 3 |
| **Avoided forest conversion** | 6,411,312 | 4,618,880 | 7 |
| **Improved forest management** | 176,059,250 | 118,859,361 | 146 |

Although not all avoided forest conversion and IFM credits are genuine removals, this breakdown illustrates both the dominance of IFM offsets in the market and the scarcity of afforestation and reforestation projects in the United States: only 3 afforestation/reforestation projects have generated credits since 2009, and the vast majority of these credits have been generated by 1 of the 3 projects. A little over 10 additional afforestation/reforestation projects are registered but have not generated any credits. Many more afforestation/reforestation projects are available internationally—the geographic scope of this report however is the United States.

## CONCLUSIONS

In conclusion, the theoretical potential to increase carbon uptake in U.S. forests is significant (100s of millions of tCO$_2$/y). However, forestry projects exhibit significant variability, depending on the type of carbon benefit (avoided loss of carbon vs. increased sequestration/removal), the accounting framework, and the time duration of the benefit. At only a few tCO$_2$ sequestered per acre per year, substantial land and transactions are required to amass sequestration levels of 1 MtCO$_2$/y or more—technology may facilitate this in the coming years.

Although measuring forest carbon and changes therein is technologically feasible today, challenges surrounding baseline definition, additionality, leakage, and durability of forestry projects are substantial and are likely to remain regardless of technological advances, absent substantial systemic reform. Even the most stringent methodologies and protocols today have come under significant criticism for over-crediting the carbon benefit of forestry offset projects, and the bulk of recently issued offsets belong to project types for which baselines, additionality, and leakage are harder to establish (e.g., IFM) and for which actual removals may represent only a portion of the credits issued.

Rather than relying on established methodologies and protocols in a blanket fashion, we recommend that corporate first movers establish their own overlay of requirements and criteria and that they screen projects individually. Given the time-consuming and expensive nature of this approach, corporate buyers interested in high-quality forestry removals may wish to also demand systemic reform and stronger common protocols.

Currently, the most robust removal opportunities in the forestry sector likely lie in applications that unambiguously increase carbon storage and trees on lands that do not have competing uses (e.g., lands that are degraded or marginal and are clearly not in agricultural production), but such land may be scarce.

# CHAPTER 5. DIRECT AIR CAPTURE

## SUMMARY

Direct Air Capture (DAC) is a straightforward method of providing negative emissions that does not suffer from questions of additionality or durability. Compared to other technology options, DAC is relatively nascent with limited global deployment and, as such, high cost. We evaluated several classes of DAC technologies in terms of their energy requirements and cost to produce $CO_2$ suitable for sequestration. All pathways have a significant energy requirement many times the thermodynamic minimum, likely leading to concurrent buildout of dedicated energy sources, particularly in the long term when the total quantity of deployed DAC capacity is large.

Areas of opportunity in research and development for all classes of DAC technology are available that will help to reduce either the cost or the uncertainty in the cost. Broadly, these research areas are related to improving the lifetime and $CO_2$ throughput of the materials, increasing the energy efficiency of processes, and reducing the cost of the raw chemicals required for consumables. In addition, due to the low total capacity of DAC currently deployed, all pathways will benefit from expanded deployment and will see cost reductions via learning-by-doing.

## KEY FINDINGS

We estimate that costs in 2030 will range from approximately $180/tCO_2$ to $450/tCO_2$ for a 1 $MtCO_2/y$ DAC facility. This cost estimate assumes improvements in sorbent lifetime and roughly 8 $MtCO_2/y$ deployed capacity for each of the individual DAC pathways, indicating that projects will need to begin breaking ground as soon as possible, with significant buildout over the next several years compared to the currently deployed capacity. At the Mt-scale, DAC facilities operating at current energy efficiencies will require a 250-MW dedicated energy source, suggesting that local energy infrastructure is a serious consideration for locating DAC facilities.

## INTRODUCTION

DAC is engineered removal of $CO_2$ from the atmosphere using purpose-built machines (*165*). Compared to many of the nature-based or mineralization processes, DAC is performed with a looping process that regenerates the sorbent or solvent used to capture the $CO_2$. This approach produces a high-concentration stream of $CO_2$ and allows the sorbent or solvent to be reused. When paired with $CO_2$ sequestration, DAC provides a route to negative emissions. DAC generally does not suffer from questions of additionality because the facilities that perform DAC are not already built, and the durability of sequestered $CO_2$ is the same as sequestered $CO_2$ from other carbon capture or BiCRS facilities. The facilities can be sited, in principle, in many locations, such as near suitable sequestration sites, though availability of energy, water, and sorbent materials may influence choice of location. Therefore, DAC represents a straightforward method of providing negative emissions. However, being a relatively nascent technology with limited global deployment, the cost of DAC remains large. In addition, the energy requirement for the regeneration part of the process can be large, which may place

constraints on the location and rate of deployment of DAC facilities, particularly since low- or zero-carbon energy sources are required for maximizing net negative emissions.

Broadly, DAC systems can be categorized based on whether they use a solid sorbent or a liquid solvent to capture $CO_2$. Solid sorbents are typically cycled between adsorption and regeneration in a temperature-, vacuum-, moisture-, or electro-swing process, for which the type of process generally indicates the mode of regeneration (Figure 5-1). For many processes, adsorption is performed by actively contacting a high–surface area sorbent with a large volume of air provided by fans, though passive air contactor designs have been proposed. The sorbents must be structured to minimize the energy required to pass air across them. As such, solid sorbent processes tend to have smaller units and can be made modular, allowing them to be mass manufactured. These modular units can be parallelized and assembled to reach the desired total capacity. Though typically operated as a cyclic semi-batch process, continuous or semi-continuous solid sorbent processes are currently under development.

In addition to the mode of regeneration, solid sorbent technologies are also distinguished by the type of adsorbent material used. Solid amine-base adsorbents (*166–169*), such as those used in the Climeworks (*170*) and Global Thermostat (*171*) processes, are typically regenerated using a temperature- and vacuum-swing process. Regeneration for these materials usually occurs between 80 and 120 °C, which means the use of waste heat sources is an option, though the heat may also be provided by a cogeneration unit. Metal-organic frameworks (MOFs) are another broad class of DAC material that can be thermally regenerated (*172–174*); the Airthena process developed by the Commonwealth Scientific and Industrial Research Organisation (CSIRO) uses electrical resistive heating to provide the thermal energy for regenerating a MOF coating (*175*). Calcium and magnesium oxides, in an enhanced weathering-like process via passive air contact, can be converted into carbonates, which can be calcined at temperatures above 900 °C to release the $CO_2$ (*176*); the Heirloom Carbon process, which uses an electric kiln to provide the high temperature, exemplifies this approach (*177*). Moisture-swing processes
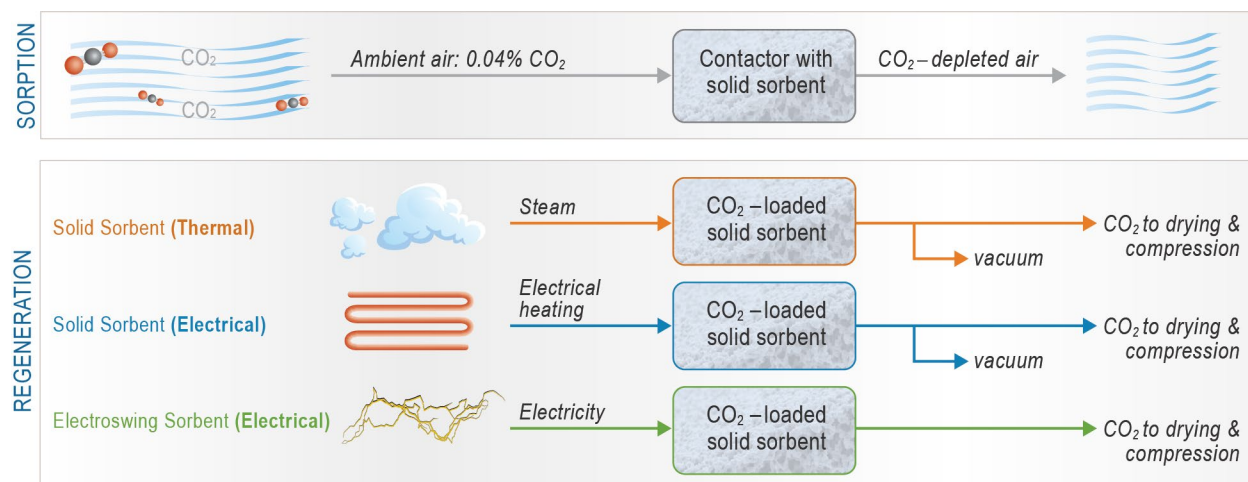


**Figure 5-1.** *Schematics for solid sorbent–based processes that cycle between adsorption and regeneration. During the sorption step, ambient air is brought in contact with the solid sorbent, which removes a portion of the CO₂. The processes considered here are distinguished by their method of regeneration. For thermally based regeneration, the energy may be provided from a waste-heat stream, directly through burning of natural gas, or via electrical resistive heating. For electroswing processes, electricity is used to directly affect the oxidation state of the sorbent.*

use ion-exchange resin-based adsorbents to capture $CO_2$ from the air. The increased humidity releases the $CO_2$; drying the adsorbent allows it to resume $CO_2$ capture (*178, 179*). Carbon Collect (formerly Silicon Kingdom Holdings) uses this process (*180*). Finally, electro-swing processes use electrical current to directly manipulate the oxidation state of an adsorbent electrode: in the charged state, the electrode binds $CO_2$; in the discharged state, the electrode releases the $CO_2$ (*181*). Verdox is developing this process.(*182*)

Liquid solvent–based processes operate in a continuous or semi-continuous manner: one part of the process contacts the liquid solvent with air, and the second part of the process pumps the $CO_2$-rich solvent to a different compartment for regeneration (Figure 5-2). Carbon Engineering developed the prototypical liquid solvent–based thermal regeneration,(*183*) which uses a potassium hydroxide solution to capture $CO_2$. The captured $CO_2$ reacts with calcium hydroxide to form calcium carbonate, which precipitates from solution and is calcined at temperatures above 900 °C to release the $CO_2$. Stoichiometric oxy-combustion of natural gas typically provides the high temperature necessary for this process, necessitating the use of an air separation unit to avoid diluting the product $CO_2$; however, the energy could also be provided via an electric kiln, similar to the Heirloom Carbon process. Amino acid–based solvents also capture $CO_2$ in a process developed by Oak Ridge National Laboratory (ORNL) (*185,186*); the $CO_2$-rich solvent reacts with guanidinium-based compounds to precipitate a carbonate and regenerate the amino acid solvent. The guanidinium carbonate salt regeneration can occur at temperatures below 120 °C to release the $CO_2$, enabling a lower temperature liquid solvent–based process.

Finally, electrodialysis and other electrochemical-based regeneration processes typically use a pH-swing to release $CO_2$ from the capture solvent by manipulating the equilibrium between gas phase $CO_2$, dissolved $CO_2$, bicarbonate ($HCO_3^-$), and carbonate ($CO_3^{2-}$). Generally, these processes use bipolar membranes to split water into acid and base, then return the base to the contactor as the capture solvent and use the acid to neutralize the $CO_2$-rich solvent, pushing the equilibrium toward gas-phase $CO_2$ (*186–188*). Other configurations use bipolar membranes in combination with anion or cation exchange membrane; $H_2$ generated at the cathode can be consumed at the anode, producing the acid required for the pH swing (*189–191*). Generally, these types of regeneration processes require large areas of membrane and highly efficient electrocatalysts to reduce the energy penalty associated with splitting water.
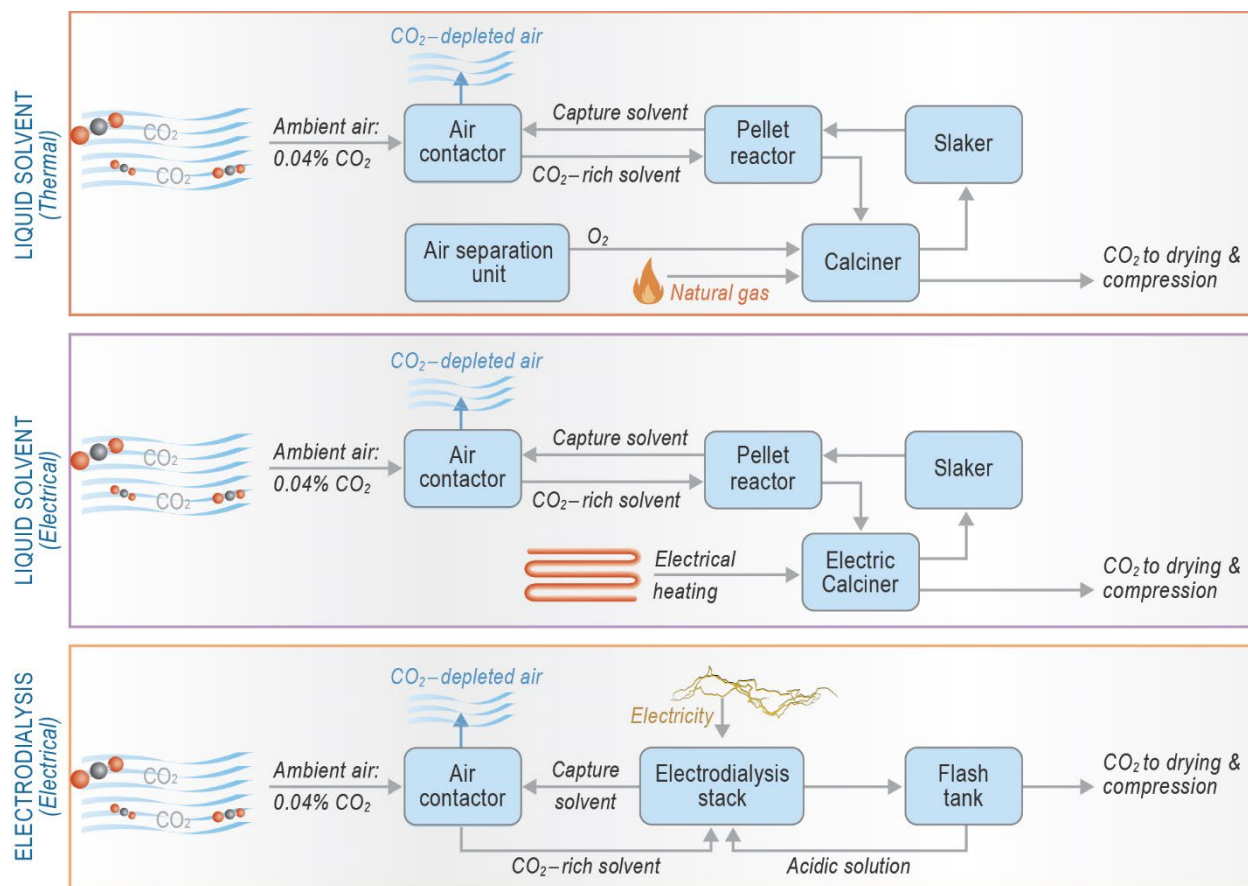
**Figure 5-2.** *Schematics for liquid solvent–based processes that operate continuously. Air is contacted with the lean capture solvent, which becomes enriched in $CO_2$. The $CO_2$-rich solvent is then pumped to the regeneration side of the process. For the prototypical liquid-solvent process, the $CO_2$ is extracted into a solid slurry, which is then calcined to release the $CO_2$. The heat for the calcination process may be provided directly through burning of natural gas or via an electric kiln. For electrodialysis processes, an electrodialysis cell is used to generate acid and base; the base returns to the air contactor, and the acid is used to control the pH of the $CO_2$-rich solvent and release the $CO_2$ from solution.*

## APPROACH AND METHODS

We evaluated these six broad classes of DAC technologies against their experimentally reported energy requirements and derived costs based on literature reports. Compared to the other negative emissions technologies considered in this report, which were evaluated for their geographic potential, DAC technologies are relatively nascent, have a low number of deployed units and a low total deployed capacity, and do not have long operating histories. Additionally, emerging technology classes that are not yet at the pilot-plant stage have the potential to be important in some geographic contexts. In the future, deployment of suitable DAC technologies should be considered in the context of proximity to suitable sequestration sites, the land available, the local climate, the local energy mix or co-building with dedicated energy production, and the availability of waste heat.

We derived the cost of net $CO_2$ removed for all processes assuming a DAC facility that separates $CO_2$ from the air and produces a gas stream at >99% $CO_2$ that is dried and compressed to 150 bar for sequestration at an average rate of 1 $MtCO_2$/y with a 90% capacity factor. We standardized the cost

evaluation framework across processes as much as possible to generally follow the methodology reported in McQueen et al. (*192*). We typically assumed the major capital equipment had a 10-year lifetime and based the economics on a 12.5% discount rate, resulting in a capital recovery factor of 18.1%. For liquid solvent processes and for the liquid contactor in the electrodialysis process, due to the larger nature of the equipment, we assumed the capital equipment had a 20-year lifetime, resulting in a capital recovery factor of 13.8%. These capital recovery factors are similar to those used in other DAC cost analyses (*193*, *194*).

For solid sorbent processes, the lifetime of the sorbent is typically much shorter than the lifetime of the DAC facility, and the sorbent will degrade with use (*195*). Therefore, we included the rate of sorbent degradation in determining the required amount of sorbent for the facility and included the annualized replacement cost for the sorbent as part of the capital cost. The sorbent was assumed to be replaced when the capacity fell to 70% of the initial capacity, the timing for which depends on the degradation rate constant and the length of the adsorption and regeneration cycle. In this report, we assumed the sorbent is cycled for approximately 8 months before it is replaced. The cost associated with replacing the sorbent more frequently to maintain a high cyclic capacity vs. operating the sorbent for longer times with fading capacity was not optimized here and is a subject for further consideration.

Finally, we added in operating expenses for thermal ($2.90/GJ for steam, $3.50/GJ for natural gas) or electrical energy ($35/MWh, $9.72/GJ for electricity via a physical power purchase agreement) and labor and maintenance to determine the total cost. Carbon emissions associated with energy production were subtracted from the 1 $MtCO_2$/y to derive the quantity of net $CO_2$ removed for each process, which was used to normalize costs. Thermal energy for the solid sorbent process was assumed to be provided by natural gas, with an emissions factor of 62.5 kg $CO_2$/GJ, and electrical energy was provided by a combination of solar and wind, with an emissions factor of 5 kg $CO_2$/GJ (*194*). Where possible, we compared our costs against costs others had reported in the literature for similar processes.

## RESULTS

Figure 5-3 shows the estimated energy requirements for the various classes of DAC technologies. The minimum thermodynamic work required to concentrate $CO_2$ from 400 ppm to >99% at the same temperature and pressure is approximately 0.5 $GJ/tCO_2$. Actual DAC processes operate far from this limit, with actual energy requirements roughly 10 to 20 times the thermodynamic minimum, though much more inefficient process configurations could be envisioned. Though not considered in this report, long-term cost forecasting with significant buildout of DAC facilities must consider the energy infrastructure available and how energy costs may change due to increased demand from DAC. At the Gt-scale, the energy demand is significant.
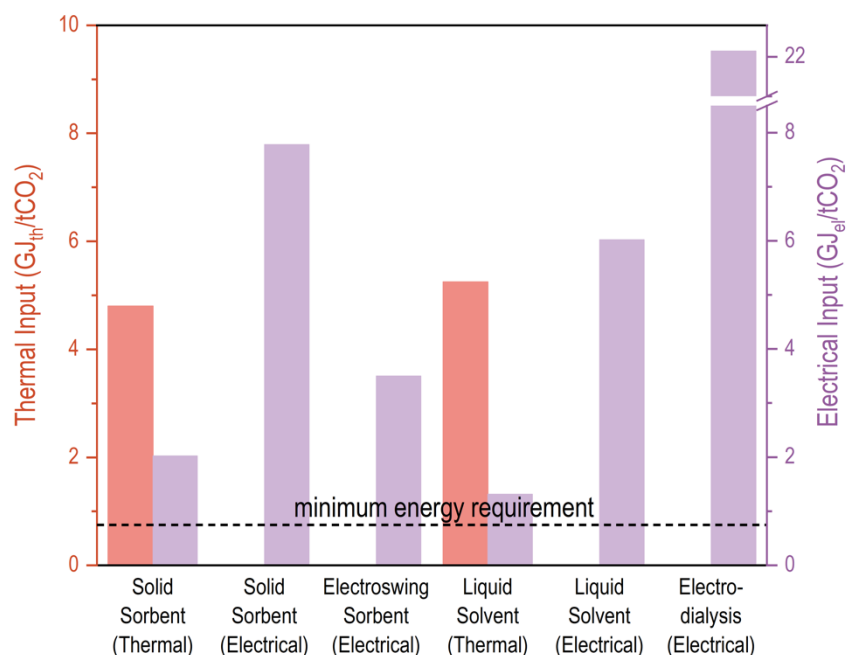
**Figure 5-3.** *Current state of required energy input to operate DAC processes and produce compressed $CO_2$ suitable for sequestration. Thermal input is typically provided via natural gas combustion but could, in principle, be provided by other sources (e.g., waste heat). For the solid sorbent (electrical) and liquid solvent (electrical) processes, electrical resistive heating is used to provide the thermal energy. For the electroswing sorbent and electrodialysis processes, electricity is used non-thermally to regenerate the material. The dashed horizontal line shows the approximate minimum energy required to concentrate $CO_2$ from the air and deliver it as a compressed gas at 150 bar.*

Many regeneration processes require heating the sorbent or solvent; in most cases, combustion of natural gas and production of steam can provide the thermal energy directly but, in principle, other sources could provide the energy (e.g., waste heat from a geothermal or nuclear power plant). In principle, electrical energy could also provide the thermal energy through resistive heating, albeit at roughly three times the cost for electricity via a physical power purchase agreement to roughly ten times the cost for commercial low-carbon electricity. For solid sorbents (e.g., amines, MOFs), resistive heating materials have been proposed to provide the relatively low temperatures (around 70–120 °C) required. For processes that require a mineral carbonation step (e.g., calcium/magnesium looping processes, with or without hydroxide solvent), large-scale calciners that can provide high temperature heat (>900 °C) are in development to replace natural gas–fired calciners.

Variants of these processes can have lower energy requirements but often at the expense of higher capital cost. For example, the electrodialysis process has a tradeoff between operating current/voltage and electrode/membrane area—resistive energy losses in the system decrease at lower current, but the material requirements increase, leading to a higher capital cost. For the particular electrodialysis process shown in DAC Figure 3, the material cost is high; thus, the overall cost of the process is reduced by minimizing the equipment size but operating with low energy efficiency/high electrical current. We explore this tradeoff more below.

The energy requirement forms the major component of the operating cost for each process, which, when combined with the annualized capital cost, provides an estimate of the total cost for a DAC

process. Figure 5-4 shows these estimates for the current cost for DAC processes, breaking the costs into rough capital/operating categories. Sorption includes equipment and operating expenses related to the ad/absorption process (e.g., fans to move air, sorbent, and solvent contactors); regeneration includes equipment and operating expenses related to releasing the $CO_2$ (e.g., calciners, steam, and electrodialysis units); and other includes labor, maintenance, and compression equipment, the latter of which is relatively constant across processes.
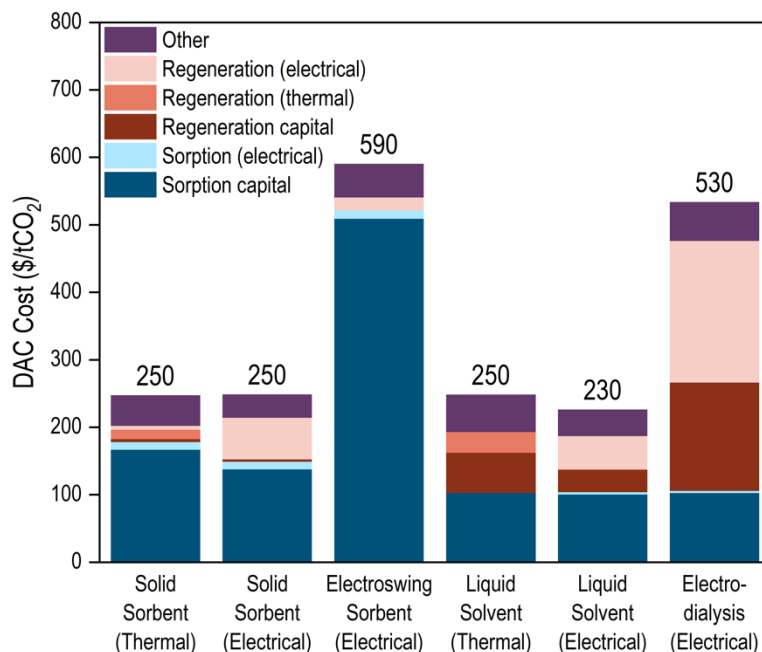


**Figure 5-4.** *Estimated current cost of net $CO_2$ removed for DAC processes at the 1-Mt$CO_2$/y facility scale.*

Generally, the costs calculated agree with the costs reported in the literature. McQueen et al. recently considered solid sorbent processes for pairing with low-carbon thermal energy (*192*). For a 100 kt$CO_2$/y facility, they estimated a cost of $223/t$CO_2$. In both cost models, the sorption capital—predominantly the replacement cost for the solid sorbent—dominates the cost. Azarabadi and Lackner identified that sorbents must typically last for tens of thousands-to-hundreds of thousands of adsorption/desorption cycles for an economically viable DAC process (*195*). Our analysis generally found this lifetime requirement to be true as well; Figure 5-5 shows the dependence of the solid sorbent process on the sorbent lifetime, which is a function of the degradation rate, the number of cycles, and the fractional $CO_2$ capacity at which the sorbent is replaced. For an adsorbent cost of $50/kg, reasonable DAC costs are only achieved as the lifetime approaches one year. This adsorbent cost is representative of the current cost of some amine-based and MOF adsorbents.
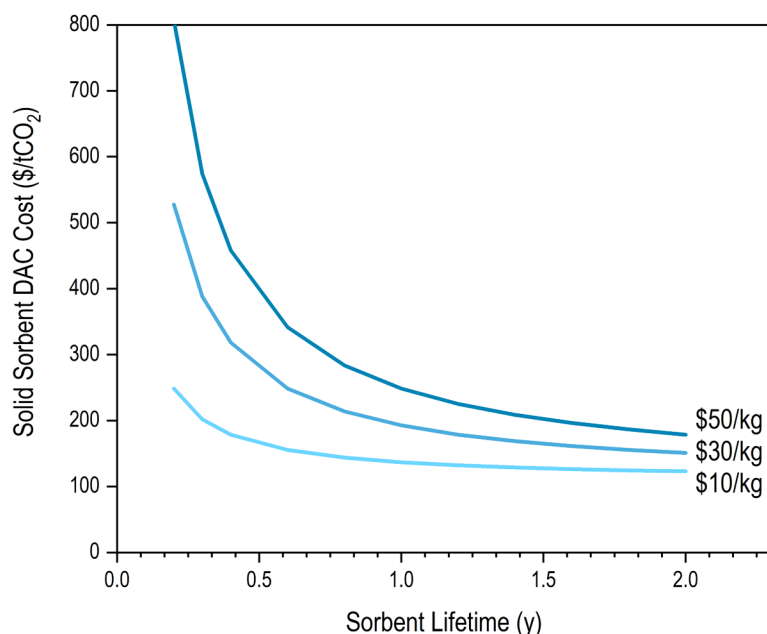
**Figure 5-5.** *The cost of solid sorbent DAC processes is highly dependent on the lifetime and cost of the sorbent. The replacement cost of the solid sorbent is indicated to the right of the lines.*

Regeneration of solid sorbent processes can also be performed using low-temperature electrical resistive heating; Sadiq et al. considered this kind of process for a MOF sorbent (*175*). In this case, electrical resistive heating is a more expensive heating method than steam but can provide a low-carbon pathway for regeneration and can allow use of adsorbents that degrade rapidly in steam.

Figure 5-5 also shows the dependence of DAC cost on the cost of the adsorbent. Assuming similar $CO_2$ throughput, very inexpensive sorbents ($10/kg) can afford to have a shorter material lifetime than more expensive sorbents. Alternatively, reductions in adsorbent cost due to increasing the scale of production, without sacrificing the quality of the material, can have significant impacts on the cost of DAC processes. The lower bounds here of $10/kg is a feasible cost target for both amine-based and MOF sorbents (*196*).

These sensitivity analyses suggest potential areas for research into reducing the cost of solid sorbent–based DAC processes: development of materials, processes, and/or process conditions that allow longer operation of the adsorbent or development of manufacturing methods that reduce the production cost of solid sorbents.

The electroswing process suffers from expensive raw materials to fabricate the electrodes, resulting in a high capital cost. In particular, the poly(anthraquinone) component of the electrode that binds the $CO_2$ has a current bulk price of $5000/kg. As with solid sorbents, the cost of production for the raw materials and the electrodes is expected to decrease due to economics of scale as the demand for the materials increases. Costs for the raw chemicals would need to decrease by one to two orders-of-magnitude for the electroswing process to be economically competitive with solid-sorbent or liquid-solvent processes.

Alternatively, other chemistries may be able to bind and release $CO_2$ with a similar electrical charging/discharging cycle and may be an area for future research.

As described by Keith et al. (*183*), a significant fraction of the cost of the thermal hydroxide solvent process is attributable to capital cost. The major pieces of equipment include the liquid-air contactor and the pellet reactor, which form the bulk of the sorption capital cost. This process includes an air separation unit to provide oxygen for firing the calciner and a power generation island to provide electricity to the rest of the process. For the electrical version of the liquid-solvent process, an electric kiln can replace the air-separation unit and power island to perform calcination. We derived the cost for the electric kiln from the National Academies report (*194*), but this cost is not well-established, as an electric kiln of this scale is not yet developed. Efficient large-scale electric kilns are a potential area for research and development, currently being undertaken by Calix (*197*).

Though electrodialysis DAC processes can have many variants on their exact membrane configuration, they all generally suffer from both high capital cost and high operating cost. The capital cost primarily comprises the membranes and electrolyzer stack hardware for electrodialysis and the liquid air contactor, which was modeled by Sabatino et al. based on the contactor in Keith et al., with some modifications due to the different optimal concentration of solvent required for the pH-swing process (*187*). In particular, the membrane cost for electrodialysis can be quite high; this process uses bipolar membranes, which are quite expensive at approximately $700/m^2$, in addition to being energy inefficient for the water-splitting reaction to produce the acid and base.

As noted previously and shown in Figure 5-6, there is a tradeoff between the operating current/voltage and the electrode/membrane area. At current electricity and membrane prices, operating with high electrical current density is more cost effective, allowing use of smaller equipment and membranes and thus reducing capital cost. However, high current density increases the energy losses due to electrical resistance and reaction overpotentials, making the process inefficient with energy usage. As membrane cost decreases, the equipment can be built larger to reduce current density and improve energy efficiency. At a membrane cost reduction of 90% of current costs, which corresponds to the ARPA-E target of $20/m^2$ for anion-exchange membranes (*198*), building out larger electrodialysis units and operating with lower current density is more cost effective. The effect of electricity cost increases is similar: more expensive electricity, which may occur in a scenario that includes a large buildout of DAC facilities, motivates use of a more energy-efficient process, though in reality a less electricity-intensive DAC process than electrodialysis may be selected.

Future research aimed at developing inexpensive, highly conductive, and selective ion-exchange membranes will improve both the capital and operating cost of electrodialysis processes. Additionally, research aimed at developing energy-efficient, noble-metal-free water-splitting catalysts will reduce the energy requirement. These research areas will likely be synergistic with existing efforts to develop membranes and catalysts for electrolyzers, fuel cells, and desalination.
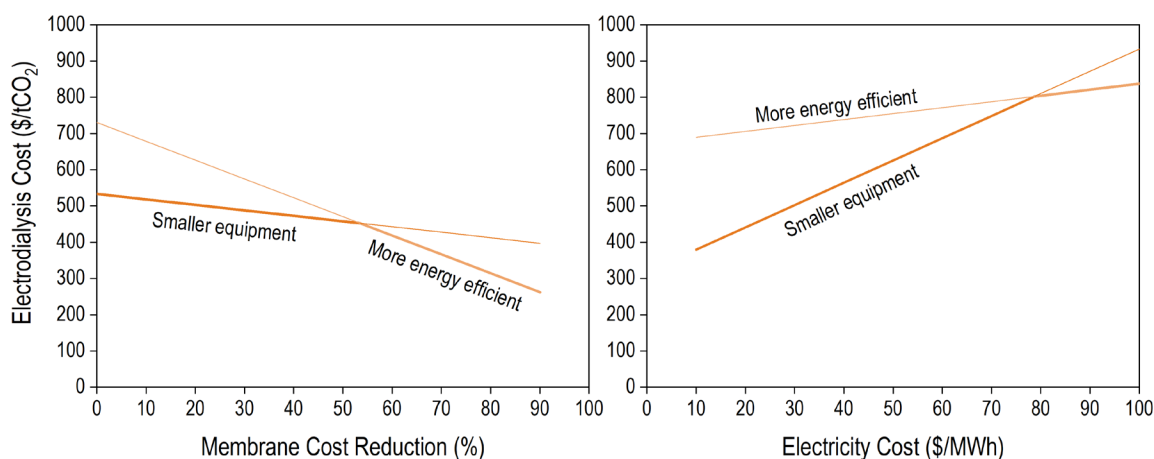
**Figure 5-6.** *The cost of electrodialysis processes is dependent on the choice of operating conditions, the cost of the ion-exchange membranes (left), and the cost of electricity (right). The costs are shown for electrodialysis processes operating at 1500 A/m² (smaller equipment) and 400 A/m² (more energy efficient, larger equipment).*

Figure 5-7 shows estimates of the cost of DAC processes in 2030. We applied several assumptions to estimate the cost reductions. For the solid sorbent processes with adsorbents that degrade, we expect the material lifetimes to improve by about 50%, which helps reduce the sorption capital cost. The unit cost of the sorbent was not reduced.

We also assumed reduced costs for the major pieces of capital equipment as a function of deployment of more DAC capacity and learning by doing (*199, 200*). We used two learning factors depending on the type of capital equipment. For small, modular equipment, such as that used in solid sorbent and electroswing processes, we applied a faster average learning rate of 15%, due to the ability to quickly learn through mass manufacturing of standardized smaller units. We also applied this learning rate to the cost of electrodes in electroswing processes and to the cost of bipolar membranes in electrodialysis processes. For larger pieces of capital equipment, such as the unique solvent contactors used in the hydroxide solvent process, we applied a more moderate average learning rate of 10%. Finally, the capital cost reduction is dependent on the total deployment of the technology, rather than on the time the technology has been operating. Therefore, an estimate of total deployed capacity in any given year is required. The Rhodium Group estimates that by 2030 total worldwide DAC capacity will be approximately 8 MtCO$_2$/y (*201*). As an optimistic estimate, we calculated costs assuming a deployed capacity of 8 MtCO$_2$/y for each of the individual DAC pathways.
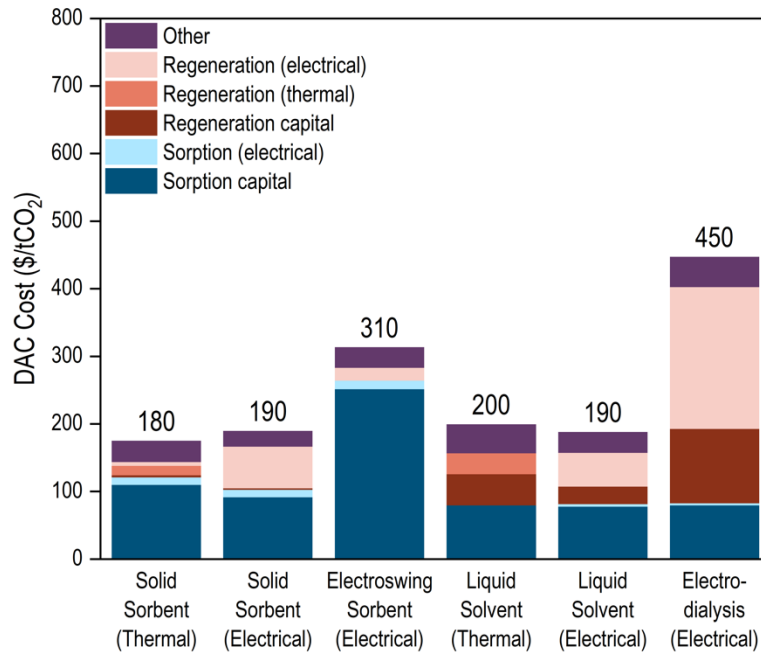
**Figure 5-7.** *Estimated cost of net $CO_2$ removed for DAC processes in 2030 assuming a capacity of 8 $MtCO_2$/y for each of the individual DAC pathways. Costs generally decrease due to improvements in solid sorbent lifetime, reduction in the cost of electrodes and membranes, and technology learning on major pieces of capital equipment.*

We did not attempt to forecast improvements in energy efficiency for any of the pathways, as these improvements will be highly dependent on the specific materials, technology, and decisions about process configuration. However, as indicated in Figure 5-3, all of the processes currently operate far from the thermodynamic minimum, indicating that potential energy-efficiency gains will be realized as research into DAC processes continues and the technology is deployed at greater total capacity and larger scale.

The impact of technology learning is largest for technologies that have significant capital cost. In particular, learning on the cost of electrodes used in the electroswing process and membranes used in the electrodialysis process, both of which are expensive components, will help to make these processes more economically competitive. However, all technologies benefit from additional deployed capacity, as can be seen in Figure 5-8, and the pace of deployment will impact the amount of technology learning that can be accomplished by 2030.

If the pace of DAC-capacity deployment is much larger than that assumed by the Rhodium Group, the cost of the DAC technologies could be even lower by 2030. For example, Heirloom Carbon has a stated goal of removing 1 $GtCO_2$/y by 2035 through their passive oxide-looping process; if the capacity of this process is assumed to double every year, their deployed capacity in 2030 would be approximately 31 $MtCO_2$/y, which would result in an additional two doublings compared to that assumed here, bringing costs down further.
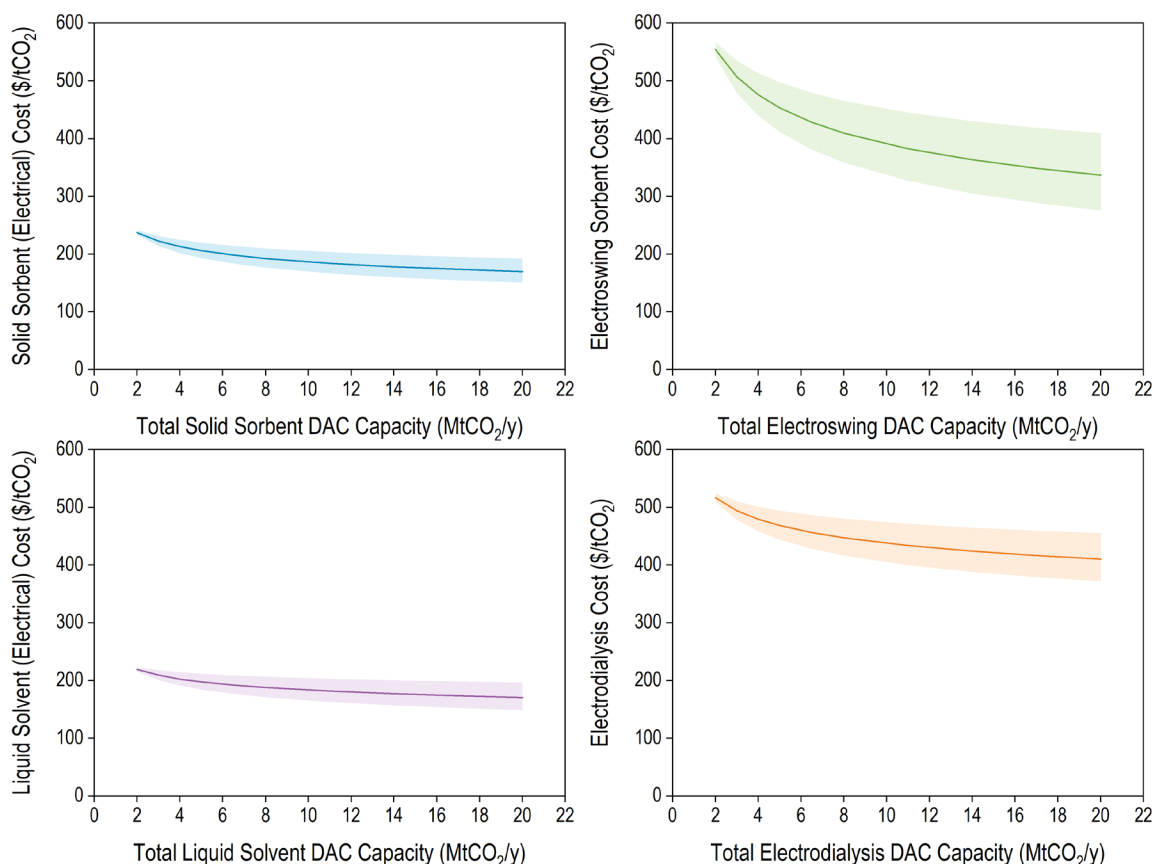
**Figure 5-8.** *Variation of DAC cost with deployed capacity. The solid line indicates the estimated cost with average learning rates of 15% for modular pieces of equipment (solid sorbent and electroswing contactors, electrodialysis membranes) and 10% for large pieces of equipment (liquid air contactors, calciners); the shaded areas indicate a range on the average learning rate of ±5%.*

## CONCLUSIONS

Overall, from this analysis, it does not appear that electroswing and electrodialysis DAC processes will overtake the current solid sorbent and liquid solvent processes in terms of cost by 2030. All technologies will benefit from getting facilities on the ground and operational; not only will this action drive down cost through traditional learning by doing, but the increased demand for sorbents and membranes may help reduce the cost of materials more quickly due to economics of scale.

Each of the specific DAC pathways has areas in which research and innovation will help drive down cost. For solid sorbent–based technologies, these areas include improving sorbent lifetime through materials and process innovations and decreasing manufacturing cost of the sorbents. For liquid solvent–based technologies, the cost of large-scale electric kilns is the major uncertainty, and their implementation will help reduce the amount of auxiliary equipment necessary for the process. Though electroswing processes are just now emerging, they hold promise if the cost of the raw chemicals for the electrodes can be reduced, either through development of new chemistries or through manufacturing innovations. Finally, electrodialysis processes will benefit from expanded research and manufacturing of ion-exchange membranes and electrocatalysts, which will be synergistic with other fields.

As noted, the energy required to run a 1 MtCO$_2$/y DAC facility will be quite large, so siting facilities with zero-carbon energy supply in mind will be critical. This need will require significant build out of additional low-carbon energy sources near suitable sequestration sites, and long-term cost projections may need to consider increased energy costs as a result of the increased demand for land. However, in the near term, opportunities exist for pairing with excess renewable electricity generation to aid in electrical grid balancing or with existing sources of waste heat from energy-production facilities.

In the long-term, many processes could become cost competitive with appropriate reductions in material costs or improvements in energy efficiency. None of the processes have inherent showstoppers that prevent them from competing with others, and exogenous factors such as local geography and climate and land and energy availability will influence which DAC technology is most appropriate for a given region.

# CHAPTER 6. CARBON MINERALIZATION

## SUMMARY

Carbon mineralization seeks to replicate and accelerate Earth's natural process of chemical weathering, which transfers $CO_2$ from the atmosphere into rocks. The removal potential of carbon mineralization is essentially unlimited given the vast quantities of suitable rocks available. While our fundamental understanding of the processes is relatively mature, practical application is lacking and field projects are a critical next step to scaling up.

This analysis considers three carbon mineralization pathways, each of which is intended to achieve negative emissions: 1) surficial carbon mineralization of existing fine-grained mine wastes (e.g., "tailings") at inactive mines only, due to concerns about the environmental and social impacts of mining and to avoid the need to offset ongoing emissions from mining activities; 2) *in situ* mineralization as geologic carbon storage, in which another $CO_2$ removal (CDR) technology, such as DAC or BiCRS, removes $CO_2$ from the atmosphere; and 3) *in situ* mineralization as standalone CDR, in which the mineralization process itself removes $CO_2$ from the atmosphere.

We developed an illustrative estimate of the cost and removal potential associated with existing mine tailings in the United States and Canada. We also qualitatively assessed the prospects for *in situ* mineralization to provide removal at scale before 2030.

## KEY FINDINGS

Asbestos tailings in the United States and Canada represent the greatest opportunity for mineralization of existing mine tailings and may have the ability to remove up to ~750 $MtCO_2$. Over 10 years, the most reactive fraction of minerals in these tailings in Canada, Vermont, and California could remove ~27, 0.4, and 0.34 $MtCO_2$ for costs of ~\$75, \$75, and \$21/$tCO_2$, respectively. However, the land use requirements to achieve this level of removal could be prohibitive—a 1-cm-thick layer of the 2 Gt of tailings in Southern Quebec would cover 76,900 $km^2$ or about 40% of the area of Washington State. **Additional interventions, such as improved grinding, mixing, and spreading, and in general better approaches than simply spreading tailings over bare ground would be needed to reduce land use.** On the order of 100–400 million tons of tailings would be required to remove Microsoft's 2030 goal of 5–6 $MtCO_2$ when relying on only the most reactive fraction of minerals, which make up ~10 wt% of suitable mine tailings at most and typically make up much less. Additional interventions, such as those listed above, would be needed to increase the reaction rate of the slower reacting but more abundant minerals.

*In situ* mineralization as storage for other CDR technologies is the most mature form of carbon mineralization and can expand the geographic range of these technologies to places where traditional saline geologic storage is not available. Reported costs of mineralization storage (~\$17–24/$tCO_2$) still exceed the average reported cost of traditional storage in sedimentary formations (~\$7–13/$tCO_2$) but should be weighed against unmonetized benefits of mineralization storage, such as the potential for greatly reduced monitoring requirements due to the high security of $CO_2$ storage once mineralization is complete. However, it is important to note this form of mineralization does not itself remove $CO_2$ from the atmosphere, it is simply another form of geologic storage.

*In situ* mineralization as a standalone CDR technology has enormous potential to scale but is immature and unlikely to provide removal at scale prior to 2030.

## INTRODUCTION

Approximately 87% of Earth's carbon is stored in rocks, which collectively contain approximately $9 \times 10^{16}$ tons of carbon (*202*). The process that incorporates carbon into rocks is called chemical weathering and it draws down about one billion tons of atmospheric $CO_2$ per year (*203*). During chemical weathering, rocks composed of minerals that contain silica, known as "silicate minerals" (e.g., olivine, pyroxene, feldspar), react with atmospheric $CO_2$ and water. The reactions split the silicate minerals, $CO_2$, and water into positively charged ions (cations), including magnesium ($Mg^{2+}$) and calcium ($Ca^{2+}$), and negatively charged ions (anions), including bicarbonate ($HCO_3^-$) and carbonate ($CO_3^{2-}$). Rivers then carry these ions to the ocean where they combine to form minerals that contain carbon, such as calcite ($CaCO_3$) and dolomite ($CaMg(CO_3)_2$), collectively known as "carbonate minerals." These carbonate minerals eventually accumulate on the ocean floor where they are formed into carbonate rocks, such as limestone. In sum, chemical weathering transfers $CO_2$ from the atmosphere to carbonate minerals, where it is stored over long timescales. It takes hundreds of thousands to millions of years for carbon to naturally cycle from the atmosphere into rocks (*203*). Carbon mineralization practices seek to speed up this natural but slow process of chemical weathering.

Rocks and minerals that are most suited for carbon mineralization are those that undergo chemical weathering most rapidly and release cations, such as magnesium and calcium, that can most readily form carbonate minerals. Such rocks include the following:

- Ultramafic igneous rocks, including dunite and peridotite, which are rocks that form in the Earth's mantle
- Mafic igneous rocks, such as basalt, which form through volcanism associated with tectonic plate boundaries and mantle hot spots
- Metamorphic rocks known as serpentinites, which form through the reaction of ultramafic igneous rocks with heat and water

There are three primary carbon mineralization pathways:

1. *In situ*: Compressed $CO_2$ or $CO_2$-saturated water is injected underground into reactive rocks
2. Surficial: Captured or ambient $CO_2$ is reacted with reactive material at or near the location of the rocks, for example mine tailings at the site of the mine or mill
3. *Ex situ*: Reactive rocks are transported to a source of $CO_2$ and/or industrial facility and reacted with captured or ambient $CO_2$, generally under high temperature and/or pressure conditions

Whether each of these pathways results in negative emissions depends on the source of $CO_2$ and the emissions associated with the carbonation process, such as energy or land-use-associated emissions. $CO_2$ used in the reaction can come from the following sources:

- Captured from a point source, such as a power plant or industrial facility
- Captured from another $CO_2$ removal technology, such a DAC or BiCRS
- Derived from ambient air or seawater

In the first pathway, any net $CO_2$ reduction would be considered avoided rather than negative emissions. The second and third pathways can generate negative emissions.

The amount of $CO_2$ that can be removed from the atmosphere through carbon mineralization has been estimated to be as much as four orders of magnitude greater than all anthropogenic $CO_2$ emissions ([16], [204])—in other words, the removal capacity is essentially unlimited given the enormous volumes and widespread geographic availability of suitable rocks.

## Current status of technology

Very few commercial carbon mineralization projects exist today. Most work on carbon mineralization to date has focused on basic or applied research at the laboratory- or small field project–scale, but practical application at the pilot-scale or larger is a rapidly growing field of research.

### In situ

To date, only two projects have demonstrated *in situ* mineralization of $CO_2$ in the field: the Wallula pilot project in Washington State and the CarbFix I pilot and CarbFix II industrial projects in Iceland.

### Wallula

The Wallula project in Washington State injected approximately 1000 tons of supercritical $CO_2$ into two permeable basalt interflow zones in the Columbia River basalt group at a depth of 830–890 m over a 3-week period in July and August 2013. Sampling and analysis conducted in the two years following injection indicated that carbonates have formed with chemical signatures consistent with the injected $CO_2$ ([205], [206]). Researchers recently conducted a new analysis of the existing data and concluded that mineralization sequestered approximately 60% of the injected $CO_2$ within two years, with the remaining $CO_2$ trapped as free-phase fluid $CO_2$. The resulting carbonates occupy approximately four percent of the available reservoir pore space, indicating that significant storage potential remains ([207]).

### CarbFix

The CarbFix projects are the most advanced carbon mineralization projects. The CarbFix I pilot project injected 175 $tCO_2$ from January to March 2012 and injected 73 tons of a 75%/25% $CO_2/H_2S$ gas mixture from the Hellisheiði geothermal plant from June to August 2012. Pilot industrial-scale operations began in 2014, simultaneously capturing $CO_2$ and $H_2S$ from two of six high-pressure turbines at the Hellisheiði geothermal plant, with plans to eventually capture virtually all emissions. To date, a total of approximately 70,000 $tCO_2$ have been injected as part of the CarbFix industrial project. CarbFix has also partnered with the Swiss DAC company, Climeworks, to store $CO_2$ captured through DAC. A small pilot DAC plant called Arctic Fox was built as part of CarbFix II. Building on the success of that pilot, the two companies built a larger combined DAC–carbon mineralization project called Orca, which commenced operations in 2021 with a capture capacity of 4000 $tCO_2/y$ ([208]).

Sampling and analysis at CarbFix indicate a complete removal of carbon in the fluids between the injection and monitoring wells, which suggests that all the $CO_2$ mineralized over this path. CarbFix estimates that greater than 95% of injected $CO_2$ mineralized in less than two years ([209–211]).

## Surficial

Several mines have considered the possibility of incorporating carbon mineralization practices into active mining operations, including diamond mines in South Africa (*212*), diamond and nickel mines and/or deposits in Canada, (*212*–*214*) and a nickel mine in Australia (*213*, *215*). Field trials are underway, but currently no commercial carbon mineralization projects are operating at active or abandoned mines.

## Ex situ

*Ex situ* projects are generally more highly engineered and require more energy inputs to pretreat the reactive material mechanically, thermally, chemically, or with a combination. The benefit of such pretreatment is significantly faster reaction rates compared to weathering at ambient conditions and the ability to achieve complete carbonation of reactive materials. However, this high level of engineering also translates into higher costs (~$50–300/tCO$_2$) that make these approaches less attractive (*214*). Newer project concepts have evolved away from strict carbon mineralization to more hybrid DAC/mineralization projects, for example the Heirloom Carbon concept, as discussed in the DAC chapter.

# Technologies considered for this report

This analysis considers three carbon mineralization technologies and/or pathways, each of which is intended to achieve negative emissions:

1. Surficial carbon mineralization of existing tailings at inactive mines only, due to Microsoft's desire to minimize the environmental and social impacts of mining and to avoid the need to offset ongoing emissions from mining activities
2. *In situ* mineralization as geologic carbon storage, in which another CDR technology, such as DAC or BiCRS, removes CO$_2$ from the atmosphere
3. *In situ* mineralization as standalone CDR, in which the mineralization process itself removes CO$_2$ from the atmosphere

The goal of the surficial pathway analysis is to provide an estimate of the cost and removal potential associated with existing mine tailings. We limited the geographic region we considered to the United States and Canada to facilitate a more thorough assessment of the opportunities in these two regions while staying within the scope of the analysis. The goal of the *in situ* pathway analysis is to provide a general summary of the prospects for *in situ* mineralization to provide removal at scale before 2030. Given this higher-level assessment, we did not apply a geographic limitation.

# ANALYSIS
## Surficial – mine tailings

### Availability

Certain mineral commodities, including asbestos, nickel (Ni), platinum-group metals (PGM), diamond, chromium (Cr), and talc, are commonly associated with ultramafic and mafic rocks, and therefore the tailings from these mining operations are generally well-suited for carbon mineralization. The United States and Canada have in the past or currently produce all these commodities.

No global database is available for existing mine tailings locations, volumes, petrology, and mineralogy. Determining the status of mines—e.g., permanently closed, temporarily suspended, active, etc.—can also be challenging. Although some resources are available, none contain all the data that would be needed to assess suitability of existing tailings for carbon mineralization (*216*). The USGS created the Mineral Resources Data System (MRDS), which is a database of global metallic and nonmetallic mineral resources (*217–220*). Available data can include deposit name, location, commodity, mine status, geologic characteristics, and production. While the MRDS contains an enormous wealth of data on mineral resources, as noted by USGS itself, the "MRDS is large, complex, and somewhat problematic." The utility of the MRDS to our analysis was limited by the availability and reliability of the data.

While the MRDS is global, it was created primarily for the United States, and coverage of the rest of the world is incomplete. The quality and completeness of records used to compile the database vary from simple reports of mineral occurrences to substantial detail on geology and extraction processes. The information most critical for our analysis included the host-rock type, mine status, and production volumes and/or mine size. Of the more than 300,000 deposits listed in the MRDS, 77.6% lack data on the host-rock type and 64.1% lack data on the production size. While the mineral commodities listed above are commonly associated with ultramafic rock types, these commodities can also occur in other rock types, so the broad lack of host-rock data limits the utility of filtering the data by commodity type. The "production size" field is only a qualitative characterization using the values Small, Medium, Large, Yes, No, and Unknown. According to the MRDS metadata, "the precise meanings of this field have changed over time and are lost to history" and whether the categories referred to production rate or cumulative production remains unclear. Only a very small subset of deposits have quantitative production data. Although all deposits have data on development status, the data are only as current as the source material from which they were derived—in other words, whether a mine is listed as active or inactive does not reflect the current day status but rather the status in the reference material used to populate the database. In addition, the USGS stopped systematically updating the MRDS in 2011. The MRDS was superseded by the USMIN mineral deposit database, which is limited to only the United States and is still under development.

We use a series of filters to attempt to identify mine sites that meet the following criteria:

- Mining occurred in the most promising rock type for carbon mineralization ("Host Rock" field = ultramafic rocks, e.g., peridotite, serpentinite, dunite, pyroxenite, etc.)
- Mines having relatively large volumes of mineral extraction and therefore likely to have relatively large volumes of tailings available ("Production Size" field = M or L)
- Mines that are inactive ("Development Status" field = Producer, Past Producer)

We combined this high-level screening with other published literature and data sources to identify the most promising sites for carbon mineralization within the constraints of this analysis. We used published estimates of tailings volumes and/or masses where possible and compared these values against tailings estimates calculated using commodity production volumes and/or masses and waste-to-product ratios. We used the latter method to calculate tailings volume estimates when published estimates were not available.

## CO$_2$ removal capacity

Rocks suitable for carbon mineralization are composed of magnesium- and iron-rich and calcium-bearing silicate minerals, including olivine, pyroxene, and serpentine minerals, and may also contain other reactive minerals like the hydroxide mineral brucite. Different rocks contain different proportions of these reactive minerals. For example, brucite typically composes only a small fraction of ultramafic rocks, up to approximately 10% by weight ("weight percent," wt%) but usually much less. Conversely, olivine is abundant, particularly in ultramafic rocks such as peridotite and dunite—a rock that is composed of greater than 90% olivine.

The total CO$_2$ that can be removed depends on the composition of the rocks and minerals that are dissolving and which minerals are forming. More specifically, the maximum CO$_2$ removal potential of any given rock or reactive material depends on the amount of Mg and Ca ions contained in the dissolving minerals and on the ratio of CO$_2$ to Mg or Ca incorporated in the carbonate minerals that form.

The CO$_2$ removal amounts listed in Table 6-1 are maximum removal amounts based on stoichiometry. The amount of removal that can be achieved under real-world conditions will be less than this theoretical maximum due to factors that affect the reaction rate and completeness, such as particle size, conditions under which the reaction is taking place (temperature, humidity, etc.), and others. Achieving complete carbonation on timescales relevant to mitigating climate change is challenging, as demonstrated by the slow natural rate of chemical weathering. Detailed site-/material-specific analysis of the mineralogy and geochemistry of reactive material is critical to determining maximum removal potential at any particular site or of any particular material. This analysis is also crucial to assessing the range of interventions or practices that may be needed to achieve complete carbonation.

**Table 6-1.** Tons of $CO_2$ sequestered per ton of dissolving and precipitating mineral. Sequestration is most efficient when dissolving minerals contain abundant Mg (e.g., brucite and forsterite) and when conditions favor formation of carbonate minerals that incorporate $CO_2$ in a 1:1 ratio with Mg (i.e., magnesite, nesquehonite, and lansfordite). Modified from (*221*)

| Tons of $CO_2$ sequestered per ton of dissolving and precipitating mineral | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| Precipitating Mineral | Name | Magnesite | Hydromagnesite | Dypingite | Pokrovskite | Artinite | Nesquehonite | Lansfordite |
| Dissolving Mineral | Formula | $MgCO_3$ | $Mg_5(CO_3)_4(OH)_2 \cdot 4H_2O$ | $Mg_5(CO_3)_4(OH)_2 \cdot {\sim}5H_2O$ | $Mg_2(CO_3)(OH)_2$ | $Mg_2(CO_3)(OH)_2 \cdot 3H_2O$ | $MgCO_3 \cdot 3H_2O$ | $MgCO_3 \cdot 5H_2O$ |
| **Name** | **Formula** | $CO_2$: MgO ratio | 1:1 | 4:5 | 4:5 | 1:2 | 1:2 | 1:1 | 1:1 |
| **Serpentine** | $[Mg_3Si_2O_5(OH)_4]$ | - | 0.48 | 0.38 | 0.38 | 0.24 | 0.24 | 0.48 | 0.48 |
| **Brucite** | $[Mg(OH)_2]$ | - | 0.75 | 0.60 | 0.60 | 0.38 | 0.38 | 0.75 | 0.75 |
| **Forsterite** | $[Mg_2SiO_4]$ | - | 0.63 | 0.50 | 0.50 | 0.31 | 0.31 | 0.63 | 0.63 |
| **Diopside** | $[CaMgSi_2O_6]$ | - | 0.41 | 0.37 | 0.37 | 0.30 | 0.30 | 0.41 | 0.41 |
| **Enstatite** | $[Mg_2Si_2O_6]$ | - | 0.44 | 0.35 | 0.35 | 0.22 | 0.22 | 0.44 | 0.44 |

*Reaction rate*

The reaction rates of the bulk rocks and constituent minerals suitable for carbon mineralization can vary over several orders of magnitude, both relative to each other and for each individual rock or mineral, depending on conditions such as pressure, temperature, and pH (*204, 222*). Brucite, fibrous serpentine minerals (e.g., chrysotile asbestos), olivine, and basaltic glass are among the fastest reacting, while pyroxenes and feldspars are slower to react. Mafic rocks are generally slower to react than ultramafic rocks because they contain higher proportions of slower reacting minerals. Research has recently focused on 1) the most reactive magnesium-bearing minerals, such as brucite, and 2) the most reactive fraction of Mg contained in various rocks and minerals, known as "labile" Mg, which makes up only 1–10 wt% of ultramafic rocks (see Box 6–1 for more information) (*223*). Rocks containing higher proportions of these highly reactive components are attractive targets for carbon mineralization.

## Box 6-1
## REACTIVE MINERALS AND LABILE MAGNESIUM

Some minerals dissolve more easily than others because of their chemical structures. For example, the silicate bonds that hold together minerals like olivine, $(Mg,Fe)_2SiO_4$, are very strong, whereas the bonds holding together a hydroxide mineral like brucite, $Mg(OH)_2$, are comparatively weak. Weakly bonded minerals dissolve more rapidly and easily release all their Mg ions, making such minerals more attractive for carbon mineralization. Some fraction of the Mg contained in reactive minerals is more readily liberated and is referred to as "labile" Mg. In silicate minerals like olivine, the labile Mg is sourced mainly from the surface of the mineral where Mg can more readily be released than from the interior of the mineral. For minerals like brucite, essentially all the Mg can be considered labile. For silicate minerals like olivine and serpentine, typically at most 10% of the Mg is labile.

Background rates of $CO_2$ uptake during silicate weathering range from 3.7–10.3 grams of $CO_2$ per square meter of rock per year ($gCO_2/m^2/y$) (*224*). Reported rates of $CO_2$-uptake from passive weathering of ultramafic mine tailings can be one to three orders of magnitude greater than this background rate (*214, 224*).

Three main factors limit the natural rate of chemical weathering: 1) rate of mineral dissolution, 2) $CO_2$ supply, and 3) rate of carbonate formation (221). Research has found that, under ambient conditions, the first two factors are most important, with carbonate formation rarely if ever being the rate-limiting factor in practice. For highly reactive phases like brucite, $CO_2$ supply is typically the rate-limiting factor in the carbonation reaction under ambient conditions, due to the low atmospheric concentration of $CO_2$ (*225*). For less reactive silicate minerals, mineral dissolution is typically the rate limiting factor under ambient conditions. Tailings management facilities are in many ways designed to prevent weathering,

which also inhibits carbonation because air and water cannot penetrate into the tailings piles. Local climate, including factors such as average temperature, humidity, rainfall, and others, also plays a significant role in reaction rate. For example, researchers estimated that the rate of carbonation at the Baptiste nickel deposit in Canada, where temperatures are below freezing for 5 out of 12 months, could be 40% lower than the carbonation rate at the Mount Keith nickel mine in Western Australia, which has a warm and dry climate (*214*).

A range of interventions can be used to speed up one or more of these rate-limiting factors and thereby increase the rate at which rocks can draw down atmospheric $CO_2$.

## *Interventions*

### Spreading, stirring, sparging

The simplest and lowest cost intervention to increase the carbonation of existing mine tailings is to increase tailings' contact with air and water, which can be accomplished by spreading the tailings into thinner layers, periodically stirring tailings piles, or injecting ("sparging") air into the interior of tailings piles. Water is critical to the carbonation reaction, and optimizing the water saturation of reactive materials creates conditions favorable to carbonate precipitation and slows the formation of passivating layers (*226*). The optimal water saturation depends on site-specific conditions, as both too little and too much water can be detrimental to the carbonation reaction.

### Grinding

The amount of reactive surface area exposed to the air has a direct influence on the carbonation rate. Some minerals, like chrysotile asbestos, naturally have a high surface area to volume ratio. Grinding reactive tailings more finely to increase the surface area, which in turn can increase the mineral dissolution rate, is therefore another intervention to increase the carbonation rate. For a given mineral or rock dissolution rate, the amount of time to achieve complete dissolution can vary from hours to years depending on the initial particle size (*17*). Another potential benefit of more finely grinding reactive material, particularly that associated with mines, is that it can improve metal recovery, which in turn can improve project economics.

### $CO_2$ supply

As noted above, for highly reactive phases like brucite, $CO_2$ supply is the rate limiting step in the carbonation reaction under ambient conditions. This limitation can be overcome by increasing the partial pressure of $CO_2$ ($pCO_2$) in the air in contact with the tailings. In laboratory experiments, increasing the $pCO_2$ from atmospheric (~0.04%) to 100% $pCO_2$ increased the reaction rate of brucite by 3 orders of magnitude (*225*). However even modest increases in $pCO_2$ can significantly speed the reaction rate. For example, the same experiment found a 2 order of magnitude increase in the carbonation rate of brucite at 10% $pCO_2$.

While, broadly speaking, the $CO_2$ supply does limit the reaction rate, research has shown that, more precisely, the limiting step is the hydration step of the $CO_2$ reaction, in which $CO_2$ dissolved in water dissociates into bicarbonate ($HCO_3^-$) and hydrogen ions ($H^+$). Interventions that can speed this step, such

as the addition of the enzyme carbonic anhydrase, could allow maximum carbonation to be achieved at $CO_2$ concentrations less than 100 $pCO_2$ (*227*).

$CO_2$ supply can be increased using emissions captured from a fossil fuel–based point source (power plant or industrial facility), but in that case the emissions reduction would be considered avoided rather than negative emissions; therefore, we have not considered this intervention for our analysis. To increase $CO_2$ supply while still achieving negative emissions, the $CO_2$ would have to be sourced from another $CO_2$ removal technology, such as DAC or BiCRS.

### *Further interventions*

Numerous additional interventions, including thermal, mechanical, and chemical pretreatment, are possible to speed the carbonation reaction but with increased costs and energy needs. These more highly engineered processes may also release more greenhouse gas (GHG) emissions than are removed, negating the benefits (*228*). As noted by Power et al., 2020 (*214*), "[a]lthough mineral carbonation was originally envisaged as an industrial process operating at elevated temperatures and pressures (e.g., 185 °C, 150 atm), the costs for pretreatment, energy use, and chemical inputs (~$50–300/t$CO_2$) far exceed current carbon prices (e.g., California Carbon Allowance ~$13 US/t)."

## *In situ* mineralization

We performed a literature review and analysis to assess the ability of *in situ* mineralization to provide removal at scale before 2030.

## RESULTS

## Surficial – mine tailings

### *Availability*

As noted above, this analysis only considers existing tailings at inactive mines, due to concerns about the environmental and social impacts of active mining operations and to avoid the need to offset ongoing emissions from mining activities. Another benefit of targeting existing tailings is that large stockpiles of appropriate material already exist, versus the relatively smaller mass of new tailings being generated at active mines. Researchers estimate that the current stockpile of ultramafic tailings is <10 Gt (an unknown fraction of which has already undergone carbon mineralization) and that new ultramafic mine tailings are produced at a rate of approximately 420 Mt/y (*223*).

### Ni, platinum group metals, and diamond mines

Ni and platinum group metals are actively mined in Canada and the United States, and diamonds are currently commercially mined in Canada. While inactive mines may also exist, the high value of these commodities could lead to inactive mines reopening, such as the Langmuir nickel mine in Ontario. Determining the current status of individual mines is also challenging, as no central database exists. The United States is a minor producer of nickel, supplying less than 1% of primary Ni production, indicating that the mass of tailings available from any inactive mines is likely relatively small. The United States is the fourth largest producer of platinum group metals, supplying less than 7% of primary palladium production, and approximately 2% of primary platinum production (*229*). Research has shown, however,

that tailings from the primary platinum group metal mine in the United States, the Stillwater Mine in Montana, are less attractive for carbon mineralization because they come primarily from the mafic section of the deposit and so contain smaller percentages of highly reactive minerals (*230*, *231*). The Stillwater Mine is also still active. While Canada is a significant producer of Ni, platinum group metals, and diamonds, indicating that significant tailings volumes may be available, as noted above these commodities are still actively mined and determining the status of individual mines (e.g., active, inactive, permanently closed) is challenging. For these reasons, we excluded all mines for these commodities from consideration.

### Chromium mines

Chromite mines occur primarily in two different types of chromite deposits: 1) stratiform, in which chromite occurs in massive seams, typically associated with layered mafic intrusions, and 2) podiform, in which chromite occurs in small magmatic bodies, typically associated with the ultramafic sections of ophiolites. The vast majority of global chromium mining occurs in stratiform deposits, such as the Bushveld Complex in South Africa and the Stillwater Mine in Montana. Podiform chromite deposits account for a very small fraction of global chromium production but constitute the entirety of inactive chromium mines in the United States and Canada that may be suitable for carbon mineralization within the constraints of our analysis. No published estimates of the volume of tailings are available.

The USGS maintains a global database of podiform chromite deposits, including estimates of reserves and production where available (*232*). In the United States, chromium was produced from a large number of small mines, primarily in California, Oregon, and at the Pennsylvania/Maryland border. The majority of mining took place prior to the 1960s, with a significant amount of mining occurring prior to 1900. Mines in California and Oregon are in very remote areas with limited access (*233*). Given these factors, the location and condition of any tailings suitable for carbon mineralization is uncertain. In Canada, podiform chromium production was concentrated in Quebec at the same mining complex as the asbestos mines (Thetford Mines), as discussed below.

Using the chromite production amounts from the USGS podiform chromite database and applying waste-to-product ratios for podiform chromite mines of 2:1 to 5:1, (*234*, *235*) we calculate that there may be on the order of 2–4 Mt of tailings in the United States and another 2–4 Mt in Canada, but this estimate is highly uncertain.

### Asbestos mines

Tailings from asbestos mining in the United States and Canada represent the greatest opportunity for carbon mineralization of mine tailings within the constraints of our analysis. Neither the United States nor Canada currently mine for asbestos, so all mines are inactive. The total amount of tailings available is uncertain but the United States may have on the order of 70 Mt of tailings and Canada may have greater than 2 Gt of tailings.

The bulk of U.S. asbestos tailings are likely split approximately equally between Vermont and California, at three mine sites. The vast majority of tailings in Canada are at the Thetford Mines mining complex in

southern Quebec. Smaller mines are also found in British Columbia, the Yukon Territory, and Newfoundland.

**Table 6-2.** U.S. and Canada Asbestos Tailings Estimates

| Country | State/province | Site | Mine(s) | Estimated tailings, Mt | Sources |
|---|---|---|---|---|---|
| **United States** | Vermont | Vermont Asbestos Group | Eden, Lowell, C-Area | 29–30 | (*230, 236*) |
| | California | New Idria/Coalinga Mining District | Union Carbide/KCAC | 0.3 | (*230, 236*) |
| | | | Johns-Manville | 1 | (*237, 238*) |
| | | | Atlas | 6 | (*230*) |
| | | Pacific Asbestos Corporation | | 30 | Calculated* |
| **Canada** | Quebec | Thetford Mines | | 2000 | (*239*) |
| | British Columbia | Cassiar | | 17–25 | (*240, 241*) |
| | Yukon Territory | Clinton Creek | | 10 | (*240*) |
| | Newfoundland | Baie | | 40 | (*241*) |

*Estimates for tailings mass at the Pacific Asbestos Corporation mine in Copperopolis, CA were extrapolated from national and California-specific asbestos production volumes, assuming a range of published waste-to-product ratios for asbestos mines (242–245). As such, these estimates are highly uncertain.*

### Removal potential

Complete carbonation of the existing asbestos tailings could sequester on the order of 700 $MtCO_2$ in Canada (*226*) and approximately 30 $MtCO_2$ in the United States. As noted above, due to the difficulty of achieving complete carbonation under ambient conditions, recent research has focused on carbonizing labile Mg and the most reactive minerals, such as brucite.

Asbestos tailings in Quebec are estimated to contain 1.8 wt% brucite, which could sequester approximately 27 $MtCO_2$ (*225*). Whether that brucite content is representative of the entire 2 Gt of available tailings remains unknown, as does how much may have reacted already. The serpentinite host rock at the New Idria Mining District in California has been estimated to contain 7–8 wt% brucite (*246*). If this reported brucite concentration is representative, the estimated ~7 Mt of tailings spread across three mines could sequester approximately 0.34 $MtCO_2$. However, how much brucite may have already reacted remains unknown. Researchers noted that brucite is absent from the top 6–9 meters of the

serpentinite host rock due to weathering and that samples of fresh serpentinite oxidized and disintegrated completely when exposed to the atmosphere for a few months due to the rapid reaction of brucite (*246*). Estimates of brucite content of tailings from the Pacific Asbestos Corporation mine are not available.

Here, we used brucite content as a proxy for labile Mg to develop a lower-bound cost estimate for carbonation of existing tailings with variable brucite content. Our analysis assumes 1) a constant $CO_2$ sequestration rate of 0.29 t$CO_2$/ton brucite/y (*225*), 2) that all tailings are exposed at the same time to conditions suitable to achieve this $CO_2$ sequestration rate, and 3) a cost of $1/ton tailings to spread tailings to the desired thickness (*223*). Published estimates for the brucite content of tailings at the Vermont Asbestos Group mine are not available so the Quebec and Vermont tailings are assumed to both have the same brucite content due to the related geologic setting (*247*).

As shown in Figure 6-1, using these simplified assumptions, removal costs for tailings in Vermont and Quebec range from approximately $75–192/t$CO_2$ removed, and for New Idria they range from $21–54/t$CO_2$, depending on how long the tailings are allowed to react. Insufficient information is available to make a cost estimate for the tailings at the Pacific Asbestos Corporation mine, but costs there will likely fall somewhere in the overall range shown of $13–345/t$CO_2$ removed, depending on labile Mg and
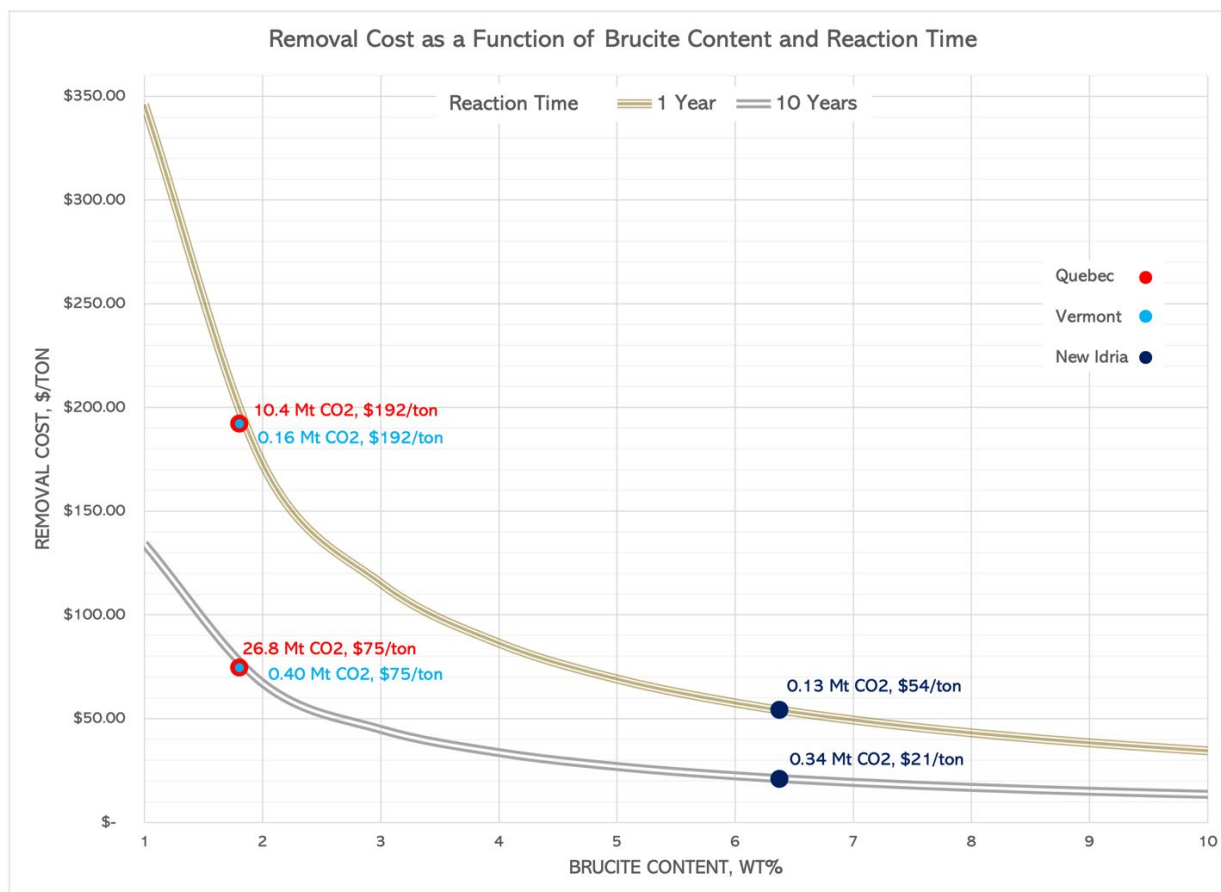


**Figure 6-1**. *CO_2 removal cost as a function of brucite content and reaction time.*

brucite content and reaction time. The $CO_2$ removal masses listed in the figure are based on the entire estimated mass of available tailings. Costs are the same regardless of the mass of tailings used.

At the brucite reaction rate specified above and assuming the rate remains constant, the brucite in all the tailings in Quebec, New Idria, and Vermont can draw down approximately 27, 0.34, and 0.4 $MtCO_2$, respectively, after 10 years of reaction. This is greater than 99% of the maximum removal capacity of all the brucite in all the tailings. The assumption of a constant reaction rate is unrealistic for such a large volume of tailings under real world conditions. In addition, some unknown fraction of the brucite in these tailings has already reacted, so the total removal capacity will be less than the maximum calculated capacity. As shown in Table 6-1 above, achieving these maximum removal capacities also requires formation of carbonates that incorporate Mg and $CO_2$ in a 1:1 ratio but, under real world conditions, a variety of different carbonates with different Mg:$CO_2$ ratios may form. Nevertheless, we provide this estimate to give a broad sense of the carbonation rate and capacity of the most reactive minerals.

Mineralization of existing mine tailings the United States and Canada could likely provide only a small fraction of Microsoft's likely demand of 5–6 $MtCO_2$ removal per year in 2030 if relying on brucite and/or labile Mg alone. To achieve that scale of removal using only the brucite in the tailings at New Idria would require more than 100 million tons of tailings—more than the entire mass available. Due to the lower brucite content in Quebec and Vermont tailings, more than 400 million tons of tailings would be required to remove 6 $MtCO_2$ using only the brucite (more than the entire mass of tailings available in Vermont). The tailings from all three locations also contain other reactive minerals. However, the minerals that make up the bulk of the tailings, primarily serpentine minerals, react much more slowly under ambient conditions. Research on tailings at the LAB-Chrysotile Mine (Thetford Mines, Canada) experimentally determined a passive carbonation rate for the bulk tailings of 4 $kgCO_2/m^3/y$ (approximately $1.5 \times 10^{-3}$ $tCO_2$/ton tailings/y), or approximately two orders of magnitude slower than the brucite carbonation rate used in the analysis above (*248*). Using this bulk carbonation rate and the same assumptions as above (i.e., constant carbonation rate, all tailings exposed to ideal conditions, $1/ton to spread tailings), the bulk tailings could sequester approximately 30 $MtCO_2$ in 10 years at a cost of approximately \$66/$tCO_2$ removed, compared to 27 $MtCO_2$ removal in 10 years at ~ \$75/$tCO_2$ for the brucite alone. In other words, highly reactive minerals and/or labile Mg are responsible for the bulk of the removal in early years.

Contrary to the simplified assumption used for our calculation, as the labile Mg reacts, the bulk carbonation rate will decrease over time to reflect the carbonation rate of the slower-reacting minerals. Achieving carbonation of this larger set of minerals over shorter timescales requires additional interventions—the exact interventions required depends on a detailed analysis of the materials and conditions at each site. The cost to spread the tailings will be the same, but additional costs will be incurred to increase the reaction rate of the slower reacting minerals. Determining and applying such interventions is critical to achieving greater levels of removal, given that labile Mg represents only a small fraction of ultramafic rocks and mine tailings. The approximately 27 Mt of carbon removal that could be achieved by the carbonation of brucite in Quebec mine tailings represents only approximately 3% of the total carbon mineralization potential of those tailings.

The land-use implications of a one-time application of tailings are significant. For example, a 1-cm-thick layer of the 2 Gt of tailings in Southern Quebec would cover 76,900 km$^2$ or about 40% of the area of Washington State. More realistic approaches include thicker piles that are stirred periodically or multiple applications of tailings over a smaller area over time. A potential drawback of this latter approach is that the slower reacting minerals may not have time to fully carbonate if they are buried beneath subsequent layers of tailings. More highly engineered methods could address this issue. For example, researchers have proposed using a series of large greenhouses where tailings could be spread in thin layers in stacked trays—both to reduce the land-use footprint and to control factors like temperature, humidity, and contact with $CO_2$—to maximize the carbonation rate (*249*).

Two of the three mine sites at the New Idria mining complex (Atlas and Johns-Manville) are designated Superfund sites, which may impact the ability to utilize their tailings. More broadly, all asbestos-containing material is subject to strict rules for handling due to the risks to human health and safety, which increases project complexity versus the use of non-hazardous feedstock materials. However, mineralization can also help reduce the health hazard associated with asbestos tailings through the conversion to carbonate minerals, providing a meaningful societal co-benefit that is not currently monetized.

The mineralized material may also have commercial value, for example in construction applications or as a cement additive (*249*). For example, the Australian company Mineral Carbonation International has developed a carbon mineralization process that includes sale of the carbonates and is currently undergoing pre-feasibility assessment by several large Australian industrial customers. Any subsequent use would have to be thoroughly evaluated to ensure it does not reverse the carbonation process and release $CO_2$.

## *In situ* mineralization

*In situ* mineralization can be divided into two broad categories: 1) $CO_2$ removal (CDR) and 2) geologic carbon storage. The key distinguishing feature between these categories is whether the mineralization process itself removes $CO_2$ from the atmosphere (1) or whether $CO_2$ is captured or removed by some other technology and then stored via carbon mineralization (2).

Type 1—CDR *in situ* mineralization (CDR-ISM)—removes $CO_2$ from the atmosphere through circulation of $CO_2$-bearing water through ultramafic rocks at depth. Type 2—storage *in situ* mineralization (Storage-ISM)—stores $CO_2$ in subsurface mafic or ultramafic rocks but, like $CO_2$ storage in sedimentary formations, does not itself remove $CO_2$ from the atmosphere. Stored $CO_2$ can be captured from point sources, such as power or industrial facilities, in which case associated emissions reductions are considered avoided emissions. Negative emissions can be achieved by pairing Storage-ISM with other CDR technologies, such as DAC or BiCRS. Regardless, Storage-ISM is simply another form of geologic carbon storage, not a negative emissions technology in and of itself.
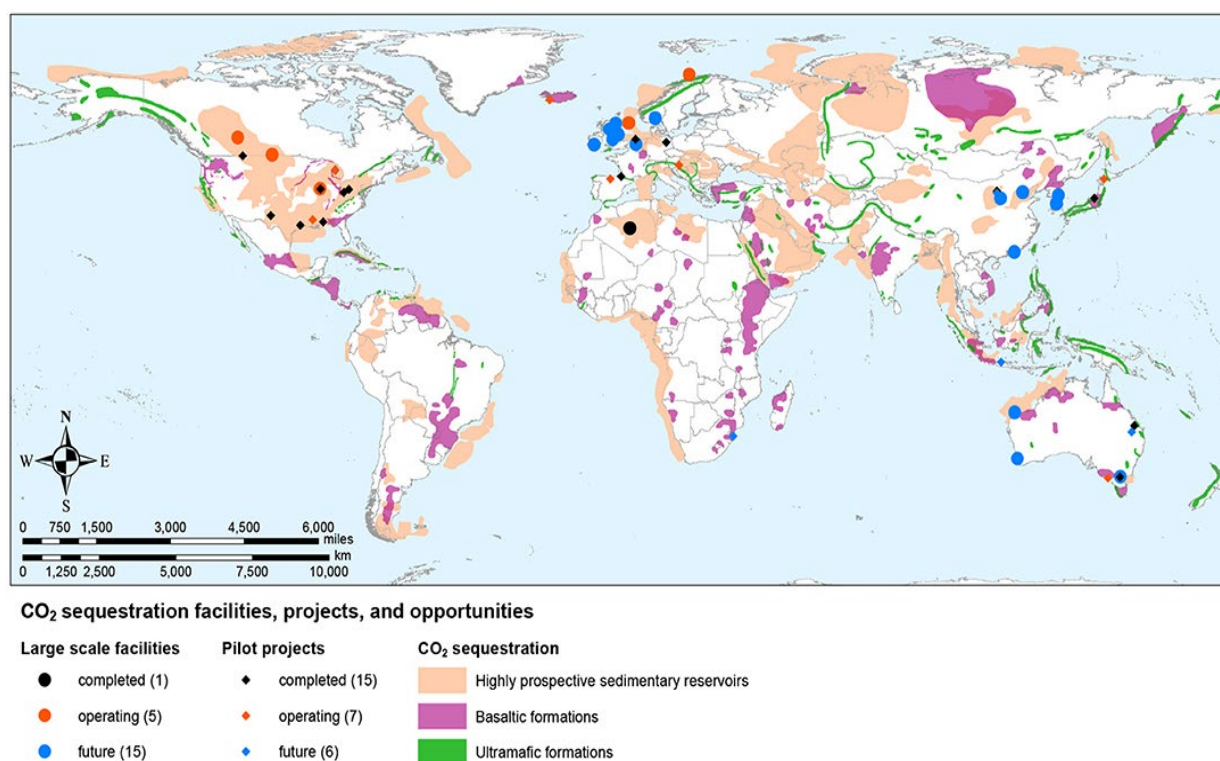
As described above, to date, only two projects have demonstrated *in situ* mineralization of $CO_2$ in the field: the Wallula pilot project in Washington state and the CarbFix I pilot and CarbFix II industrial

projects in Iceland, both of which are Storage-ISM projects. Although they generate negative emissions, the Arctic Fox and Orca combined DAC-mineralization projects run by Climeworks and CarbFix are also Storage-ISM projects because the mineralization process itself is not removing $CO_2$ from the atmosphere, DAC is. No CDR-ISM field projects—in which mineralization itself is removing $CO_2$ from the atmosphere—currently exist. As such, much remains unknown about how *in situ* carbon mineralization can be safely implemented on a scale that will meaningfully contribute to climate change mitigation. These existing projects and studies of natural analogues have helped constrain the risks and uncertainties associated with *in situ* mineralization and have generated improved strategies to achieve safe and rapid mineralization, but uncertainties remain regarding the ability to significantly scale up project size.

### *Storage-In Situ Mineralization*

CarbFix reports costs of \$24.8/$tCO_2$ for their carbon capture and storage project at the Hellisheiði geothermal power plant (*208*). CarbFix also recently announced the creation of Coda Terminal, a storage hub in Iceland to accept $CO_2$ from third parties and store it via carbon mineralization and indicated that it anticipates storage costs of approximately €15/ton (*250*). These costs are approximately double the average reported cost to store $CO_2$ in sedimentary formations in the United States (\$7–13/$tCO_2$), although they are within the range of U.S. saline storage costs (*251–253*).

The higher cost relative to saline storage must be weighed against the greater storage security. For this reason, in some cases Storage-ISM may be preferable even in places where saline storage is available. Ongoing monitoring compliance costs may be lower for Storage-ISM, which could make it more cost competitive with saline storage. As noted by Blondes et al., 2018 (*253*), some saline storage projects may also need to manage pressure build-up by either reducing injection volumes or producing brine, which could increase saline storage costs to \$20–80/$tCO_2$ or more.

**CO₂ sequestration facilities, projects, and opportunities**

**Large scale facilities**
- ● completed (1)
- ● operating (5)
- ● future (15)

**Pilot projects**
- ◆ completed (15)
- ◆ operating (7)
- ◆ future (6)

**CO₂ sequestration**
- Highly prospective sedimentary reservoirs
- Basaltic formations
- Ultramafic formations

**Figure 6-2.** *Map showing the location of onshore basaltic (purple) and ultramafic (green) rocks suitable for carbon mineralization. From (204)*

Storage-ISM paired with DAC or BECCS/BiCRS could expand the geographic range for these projects to places where saline storage is not available. As shown in Figure 6-2, locations with geology appropriate for saline storage overlap little with geology appropriate for *in situ* mineralization. In locations where both types of storage are available, Storage-ISM may also be preferable to saline storage where greater storage security is desired.

While results from field projects to date are promising, additional research is needed to resolve questions around scalability, efficiency, monitoring, accounting, and other issues before *in situ* mineralization can provide storage at scale.

### CDR-In Situ Mineralization

Water at the Earth's surface naturally contains dissolved $CO_2$ derived from the atmosphere. When this water circulates underground through ultramafic rocks like peridotite, the $CO_2$ in the water reacts with minerals in the ultramafic rock to form new carbonate minerals. When returned to the surface, the $CO_2$-depleted water then absorbs more atmospheric $CO_2$, and the process is repeated. This natural process can be replicated and accelerated to increase the rate of atmospheric carbon removal by drilling a series of wells to increase the rate of water circulation and $CO_2$ removal (223). If the amount of $CO_2$ removed is greater than any emissions associated with the process, negative emissions can be achieved.

*In situ* mineralization as a standalone carbon removal strategy has enormous ability to scale, potentially at similar or lower cost, energy use, and land use than surficial or *ex situ* mineralization or DAC (223).

The volume of suitable rock is vast, but much is in difficult-to-access places, such as offshore. To date, no real-world projects exist, and many fundamental research questions remain.

Although the two practices have key differences, considering the timeline associated with scaling Storage-ISM as a proxy for the potential development timeline for CDR-ISM may prove instructive. As noted above, two field projects exist to date, with the CarbFix project being the most advanced. Approximately 15 years have lapsed from the start of site characterization work to the current level of development, in which approximately 70,000 $tCO_2$ captured from the geothermal plant have been injected, and CarbFix has begun injection of an estimated 4000 $tCO_2$/y captured from the new Climeworks DAC facility (Orca). Having multiple field projects operating could potentially shorten the time to commercialization for CDR-ISM relative to CarbFix but, to maximize the application of learnings, projects should be staggered rather than commencing simultaneously.

Readily accessible sites should be prioritized for small-scale field testing (i.e., similar in size to Wallula and CarbFix I). A coordinated network of field sites, similar in concept to the Regional Carbon Sequestration Partnerships, could be used to investigate key research questions more efficiently. Given the current low level of technological readiness and the remaining fundamental research questions, we do not anticipate that CDR-ISM will provide removal at scale in the next decade.

## CONCLUSIONS

Asbestos tailings in the United States and Canada represent the best opportunity for carbon removal using existing mine tailings. Additional, site-specific research is needed to constrain the mineralization potential and identify interventions that can optimize mineralization efficiency and rate. While using asbestos-containing material increases the regulatory complexity, it also provides a critical human-health benefit by helping remediate the threats associated with the material. Medium- to large-scale field trials are a critical next step to scaling this removal option.

*In situ* mineralization as storage for other CDR technologies, like DAC and BiCRS, is the most mature of the three pathways assessed here. While not a negative emissions technology in and of itself, *in situ* mineralization as storage is an important enabling technology for other CDR pathways and offers an alternative to traditional saline geologic storage where such storage is not available or where a shortened timeframe to monitor for storage security is desired. The current higher average cost of mineralization storage should be weighed against these potential benefits.

*In situ* mineralization as a standalone CDR pathway is promising but immature and is unlikely to provide removal at scale prior to 2030.

# CHAPTER 7. REFERENCES

1.  S. Osaka, R. Bellamy, N. Castree, Framing "nature-based" solutions to climate change. *WIREs Climate Change*. **12**, e729 (2021).

2.  Carbon Direct, Microsoft, "Criteria for high-quality carbon dioxide removal" (2021), (available at https://query.prod.cms.rt.microsoft.com/cms/api/am/binary/RWGG6f).

3.  R. Gross, R. Hanna, A. Gambhir, P. Heptonstall, J. Speirs, How long does innovation and commercialisation in the energy sectors take? Historical case studies of the timescale from invention to widespread commercialisation in energy supply and end use technology. *Energy Policy*. **123**, 682–699 (2018).

4.  David Sandalow, Roger Aines, Julio Friedman, Colin McCormick, Daniel L. Sanchez, "Biomass Carbon Removal and Storage Roadmap," *Cool Earth* (LLNL-TR-815200 https://www.icef.go.jp/pdf/summary/roadmap/icef2020_roadmap.pdf, Ministry of Economy, Trade, and Industry, Japan, 2021), (available at https://www.icef.go.jp/pdf/summary/roadmap/icef2020_roadmap.pdf).

5.  Sarah E. Baker, Joshuah K. Stolaroff, George Peridas, Simon H. Pang, Hannah M. Goldstein, Felicia R. Lucci, Wenqin Li, Eric W. Slesserev, Jennifer Pett-Ridge, Frederick J. Ryerson, Jeff L. Wagoner, Whitney Kirkendall, Roger D. Aines, Daniel L. Sanchez, Bodie Cabiyo, Joffre Baker, Sean McCoy, Sam Uden, Ron Runnenbaum, Jennifer Wilcox, Peter C. Psarras, Helene Pilorge, Noah McQueen, Daniel Maynard, Colin McCormick, "Getting to Neutral: Options for Negative Carbon Emissions in California" (LLNL-TR-796100, Lawrence Livermore National Laboratory, 2020).

6.  J. Sanderman, T. Hengl, G. J. Fiske, Soil carbon debt of 12,000 years of human land use. **113**, 9575–9580 (2017).

7.  A. Mayer, Z. Hausfather, A. D. Jones, W. L. Silver, The potential of agricultural land management to contribute to lower global surface temperatures. *Sci. Adv.* **4**, eaaq0932 (2018).

8.  B. Minasny, B. P. Malone, A. B. McBratney, D. A. Angers, D. Arrouays, A. Chambers, V. Chaplot, Z.-S. Chen, K. Cheng, B. S. Das, D. J. Field, A. Gimona, C. B. Hedley, S. Y. Hong, B. Mandal, B. P. Marchant, M. Martin, B. G. McConkey, V. L. Mulder, S. O'Rourke, A. C. Richer-de-Forges, I. Odeh, J. Padarian, K. Paustian, G. Pan, L. Poggio, I. Savin, V. Stolbovoy, U. Stockmann, Y. Sulaeman, C.-C. Tsui, T.-G. Vågen, B. van Wesemael, L. Winowiecki, Soil carbon 4 per mille. *Geoderma*. **292**, 59–86 (2017).

9.  K. Paustian, J. Lehmann, S. Ogle, D. Reay, G. P. Robertson, P. Smith, Climate-smart soils. *Nature*. **532**, 49–57 (2016).

10. A. Swan, S. A. Williams, K. Brown, A. Chambers, J. Creque, J. Wick, K. Paustian, Carbon and Greenhouse Gas Evaluation for NRCS Conservation Practice Planning, 68 (2019).

11. P. Eavis, C. Krauss, What's Really Behind Corporate Promises on Climate Change? *The New York Times* (2021), (available at https://www.nytimes.com/2021/02/22/business/energy-environment/corporations-climate-change.html).

12.    Science Based Targets Initiative, "From ambition to impact: Science Based Targets Initiative Annual Progress Report, 2020" (2021), (available at https://sciencebasedtargets.org/resources/files/SBTiProgressReport2020.pdf).

13.    Brad Smith, Microsoft will be carbon negative by 2030. *The Official Microsoft Blog* (2020), (available at https://blogs.microsoft.com/blog/2020/01/16/microsoft-will-be-carbon-negative-by-2030/).

14.    IPCC, "Summary for Policymakers. In: Global Warming of 1.5°C. An IPCC Special Report on the impacts of global warming of 1.5°C above pre-industrial levels and related global greenhouse gas emission pathways, in the context of strengthening the global response to the threat of climate change, sustainable development, and efforts to eradicate poverty [Masson-Delmotte, V., P. Zhai, H.-O. Pörtner, D. Roberts, J. Skea, P.R. Shukla, A. Pirani, W. Moufouma-Okia, C. Péan, R. Pidcock, S. Connors, J.B.R. Matthews, Y. Chen, X. Zhou, M.I. Gomis, E. Lonnoy, T. Maycock, M. Tignor, and T. Waterfield (eds.)]." (In Press, 2018).

15.    J. Rogelj, A. Popp, K. V. Calvin, G. Luderer, J. Emmerling, D. Gernaat, S. Fujimori, J. Strefler, T. Hasegawa, G. Marangoni, V. Krey, E. Kriegler, K. Riahi, D. P. van Vuuren, J. Doelman, L. Drouet, J. Edmonds, O. Fricko, M. Harmsen, P. Havlík, F. Humpenöder, E. Stehfest, M. Tavoni, Scenarios towards limiting global mean temperature increase below 1.5 °C. *Nature Clim Change*. **8**, 325–332 (2018).

16.    IPCC, "Summary for Policymakers. In: Climate Change 2021: The Physical Science Basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change [Masson-Delmotte, V., P. Zhai, A. Pirani, S. L. Connors, C. Péan, S. Berger, N. Caud, Y. Chen, L. Goldfarb, M. I. Gomis, M. Huang, K. Leitzell, E. Lonnoy, J.B.R. Matthews, T. K. Maycock, T. Waterfield, O. Yelekçi, R. Yu and B. Zhou (eds.)]." (In Press, Cambridge University Press, 2021).

17.    National Academies of Sciences, Engineering, and Medicine, *Negative Emissions Technologies and Reliable Sequestration: A Research Agenda* (National Academies Press, Washington, D.C., 2019; https://doi.org/10.17226/25259.).

18.    V. Masson-Delmotte, P. Zhai, H. O. Portner, D. Roberts, J. Skea, P. R. Shukla, A. Pirani, W. Moufouma-Okia, C. Pean, R. Pidcock, S. Connors, J. B. R. Matthews, Y. Chen, X. Zhou, M. I. Gomis, E. Lonnoy, T. Maycock, M. Tignor, T. Waterfield, "Summary for Policymakers, in Global Warming of 1.5C: An IPCC Special Report on the impacts of global warming of 1.5C above pre-industrial levels and related global greenhouse gas emission pathways, in the contect of strengthening the global response to the threat of climate change, sustainable development, and efforts to eradicate poverty." (IPCC, 2018, IPCC).

19.    M. Bui, D. Zhang, M. Fajardy, N. Mac Dowell, Delivering carbon negative electricity, heat and hydrogen with BECCS – Comparing the options. *International Journal of Hydrogen Energy*. **46**, 15298–15321 (2021).

20.    C. Consoli, "Bioenergy and Carbon Capture and Storage" (2019), (available at https://www.globalccsinstitute.com/wp-content/uploads/2019/03/BECCS-Perspective_FINAL_PDF.pdf).

21. ADM announces successful completion of 1 million metric ton capture and storage project (2021), (available at https://www.adm.com/news/news-releases/adm-announces-successful-completion-of-one-million-metric-ton-carbon-capture-and-storage-project).

22. I. Butnar, O. Broad, B. Solano Rodriguez, P. E. Dodds, The role of bioenergy for global deep decarbonization: $CO_2$ removal or low-carbon energy? *GCB Bioenergy*. **12**, 198–212 (2020).

23. S. Fuss, F. Johnsson, The BECCS Implementation Gap–A Swedish Case Study. *Front. Energy Res.* **8**, 553400 (2021).

24. D. Tilman, R. Socolow, J. A. Foley, J. Hill, E. Larson, L. Lynd, S. Pacala, J. Reilly, T. Searchinger, C. Somerville, R. Williams, Beneficial Biofuels--The Food, Energy, and Environment Trilemma. *Science*. **325**, 270–271 (2009).

25. O. Englund, P. Börjesson, G. Berndes, N. Scarlat, J.-F. Dallemand, B. Grizzetti, I. Dimitriou, B. Mola-Yudego, F. Fahl, Beneficial land use change: Strategic expansion of new biomass plantations can reduce environmental impacts from EU agriculture. *Global Environmental Change*. **60**, 101990 (2020).

26. K. L. Kline, S. Msangi, V. H. Dale, J. Woods, G. M. Souza, P. Osseweijer, J. S. Clancy, J. A. Hilbert, F. X. Johnson, P. C. McDonnell, H. K. Mugera, Reconciling food security and bioenergy: priorities for action. *GCB Bioenergy*. **9**, 557–576 (2017).

27. H. I. Jager, E. S. Parish, M. H. Langholtz, A. W. King, Perennials in Flood-Prone Areas of Agricultural Landscapes: A Climate Adaptation Strategy. *BioScience*. **70**, 278–280 (2020).

28. Gas Technology Institute, "Low Carbon Renewable Natural Gas from Wood Wastes" (Gas Technology Institute, 2019), (available at https://www.gti.energy/wp-content/uploads/2019/02/Low-Carbon-Renewable-Natural-Gas-RNG-from-Wood-Wastes-Final-Report-Feb2019.pdf.

29. J. Riaza, M. V. Gil, L. Álvarez, C. Pevida, J. J. Pis, F. Rubiera, Oxy-fuel combustion of coal and biomass blends. *Energy*. **41**, 429–435 (2012).

30. M. B. Toftegaard, J. Brix, P. A. Jensen, P. Glarborg, A. D. Jensen, Oxy-fuel combustion of solid fuels. *Progress in Energy and Combustion Science*. **36**, 581–625 (2010).

31. F. Sher, M. A. Pans, C. Sun, C. Snape, H. Liu, Oxy-fuel combustion study of biomass fuels in a 20 kWth fluidized bed combustor. *Fuel*. **215**, 778–786 (2018).

32. P. Wienchol, A. Szlęk, M. Ditaranto, Waste-to-energy technology integrated with carbon capture – Challenges and opportunities. *Energy*. **198**, 117352 (2020).

33. M. H. Langholtz, B. J. Stokes, L. M. Eaton, "2016 Billion-Ton Report: Advancing Domestic Resources for a Thriving Bioeconomy" (DOE/EE-1440, ORNL/TM-2016/160, 1271651, 2016), p. DOE/EE-1440, ORNL/TM-2016/160, 1271651, , doi:10.2172/1271651.

34. Matthew Langholtz, ORNL, Personal Communication (2020).

35.     Anelia Milbrandt, NREL, Personal Communication (2020).

36.     E. Baik, D. L. Sanchez, P. A. Turner, K. J. Mach, C. B. Field, S. M. Benson, Geospatial analysis of near-term potential for carbon-negative bioenergy in the United States. *Proc Natl Acad Sci USA*. **115**, 3290–3295 (2018).

37.     M. Langholtz, I. Busch, A. Kasturi, M. R. Hilliard, J. McFarlane, C. Tsouris, S. Mukherjee, O. A. Omitaomu, S. M. Kotikot, M. R. Allen-Dumas, C. R. DeRolph, M. R. Davis, E. S. Parish, The Economic Accessibility of CO2 Sequestration through Bioenergy with Carbon Capture and Storage (BECCS) in the US. *Land*. **9**, 299 (2020).

38.     J. K. Stolaroff, S. H. Pang, W. Li, W. G. Kirkendall, H. M. Goldstein, R. D. Aines, S. E. Baker, Transport Cost for Carbon Removal Projects With Biomass and CO2 Storage. *Front. Energy Res.* **9**, 639943 (2021).

39.     G. Liu, E. D. Larson, R. H. Williams, T. G. Kreutz, X. Guo, Making Fischer–Tropsch Fuels and Electricity from Coal and Biomass: Performance and Cost Analysis. *Energy Fuels*. **25**, 415–437 (2011).

40.     E. D. Larson, H. Jin, F. E. Celik, Large-scale gasification-based coproduction of fuels and electricity from switchgrass. *Biofuels, Bioprod. Bioref.* **3**, 174–194 (2009).

41.     A. Zoelle, D. Keairns, L. L. Pinkerton, M. J. Turner, M. Woods, N. Kuehn, V. Shah, V. Chou, "Cost and Performance Baseline for Fossil Energy Plants Volume 1a: Bituminous Coal (PC) and Natural Gas to Electricity Revision 3" (DOE/NETL-2015/1723, 2015), (available at https://www.netl.doe.gov/energy-analysis/details?id=729).

42.     E. Connelly, M. Penev, A. Elgowainy, C. Hunter, "Current Status of Hydrogen Liquefaction Costs" (DOE Hydrogen and Fuel Cells Program Record 19001, 2019), (available at https://www.hydrogen.energy.gov/pdfs/19001_hydrogen_liquefaction_costs.pdf).

43.     W. A. Amos, "Costs of Storing and Transporting Hydrogen" (NREL/TP-570-25106, 1999), (available at https://www.osti.gov/biblio/6574-costs-storing-transporting-hydrogen).

44.     M. J. Turner, L. L. Pinkerton, "Quality Guidelines for Energy System Studies: Capital Cost Scaling Methodology" (DOE/NETL--341/013113, 1513277, 2013), p. DOE/NETL--341/013113, 1513277, , doi:10.2172/1513277.

45.     Timothy Grant, "Carbon Dioxide Transport and Storage Costs in NETL Studies" (DOE/NETL-2017/1819, National Energy Technology Laboratory, 2017), (available at https://netl.doe.gov/projects/files/QGESSCarbonDioxideTransportandStorageCostsinNETLStudies _110617.pdf).

46.     O. US EPA, US EPA Emissions & Generation Resource Integrated Database (eGRID) (2020), (available at https://www.epa.gov/egrid/data-explorer).

47.     W. J. Sagues, H. Jameel, D. L. Sanchez, S. Park, Prospects for bioenergy with carbon capture & storage (BECCS) in the United States pulp and paper industry. *Energy & Environmental Science*. **13**, 2243–2261 (2020).

48. Carnegie Mellon University, National Energy Technology Laboratory, Integrated Environmental Control Model, (available at https://www.cmu.edu/epp/iecm/).

49. H. Zhai, Y. Ou, E. S. Rubin, Opportunities for Decarbonizing Existing U.S. Coal-Fired Power Plants via CO2 Capture, Utilization and Storage. *Environ. Sci. Technol.* **49**, 7571–7579 (2015).

50. P. R. R. Rochedo, I. V. L. Costa, M. Império, B. S. Hoffmann, P. R. de C. Merschmann, C. C. N. Oliveira, A. Szklo, R. Schaeffer, Carbon capture potential and costs in Brazil. *Journal of Cleaner Production*. **131**, 280–295 (2016).

51. B. O. Ugwuishiwu, J. N. Nwakaire, C. J. Ohagwu, Cost analysis of carbon capture and storage for current gas-fired power plants in Nigeria. *Greenhouse Gases: Science and Technology*. **9**, 370–386 (2019).

52. H. Gerbelová, M. van der Spek, W. Schakel, Feasibility Assessment of CO2 Capture Retrofitted to an Existing Cement Plant: Post-combustion vs. Oxy-fuel Combustion Technology. *Energy Procedia*. **114**, 6141–6149 (2017).

53. R. Siddique, Utilization of municipal solid waste (MSW) ash in cement and mortar. *Resources, Conservation and Recycling*. **54**, 1037–1047 (2010).

54. A. Bosmans, I. Vanderreydt, D. Geysen, L. Helsen, The crucial role of Waste-to-Energy technologies in enhanced landfill mining: a technology review. *Journal of Cleaner Production*. **55**, 10–23 (2013).

55. M. Bui, M. Fajardy, N. Mac Dowell, Bio-energy with carbon capture and storage (BECCS): Opportunities for performance improvement. *Fuel*. **213**, 164–175 (2018).

56. N. Pour, P. A. Webley, P. J. Cook, Potential for using municipal solid waste as a resource for bioenergy with carbon capture and storage (BECCS). *International Journal of Greenhouse Gas Control*. **68**, 1–15 (2018).

57. Anthony Turhollow, Erin Wilkerson, Shahab Sokhansanj, "Cost Methodology for Biomass Feedstocks: Herbaceous Crops and Agricultural Residues" (ORNL/TM-2008/105, 2009), (available at https://info.ornl.gov/sites/publications/files/Pub11927.pdf).

58. K. Gustafsson, R. Sadegh-Vaziri, S. Grönkvist, F. Levihn, C. Sundberg, BECCS with combined heat and power: Assessing the energy penalty. *International Journal of Greenhouse Gas Control*. **108**, 103248 (2021).

59. G. R. Sanford, J. L. Posner, R. D. Jackson, C. J. Kucharik, J. L. Hedtcke, T.-L. Lin, Soil carbon lost from Mollisols of the North Central U.S.A. with 20 years of agricultural best management practices. *Agriculture, Ecosystems & Environment*. **162**, 68–76 (2012).

60. D. P. McLaren, D. P. Tyfield, R. Willis, B. Szerszynski, N. O. Markusson, Beyond "Net-Zero": A Case for Separate Targets for Emissions Reduction and Negative Emissions. *Front. Clim.* **1**, 4 (2019).

61.  E. E. Oldfield, A. J. Eagle, R. L. Rubin, J. Sanderman, D. R. Gordon, Agricultural soil carbon credits: Making sense of protocols for carbon sequestration and net greenhouse gas removals, (available at edf.org/sites/default/files/content/agricultural-soil-carbon-credits-protocolsynthesis. pdf).

62.  J. Sanderman, J. A. Baldock, Accounting for soil carbon sequestration in national inventories: a soil scientist's perspective. *Environ. Res. Lett.* **5**, 034003 (2010).

63.  K. R. Olson, M. Al-Kaisi, R. Lal, B. Lowery, Examining the paired comparison method approach for determining soil organic carbon sequestration rates. *Journal of Soil and Water Conservation*. **69**, 193A-197A (2014).

64.  K. R. Olson, M. M. Al-Kaisi, R. Lal, B. Lowery, Experimental Consideration, Treatments, and Methods in Determining Soil Organic Carbon Sequestration Rates. *Soil Science Society of America Journal*. **78**, 348–360 (2014).

65.  J. P. Mitchell, A. Shrestha, K. Mathesius, K. M. Scow, R. J. Southard, R. L. Haney, R. Schmidt, D. S. Munk, W. R. Horwath, Cover cropping and no-tillage improve soil health in an arid irrigated cropping system in California's San Joaquin Valley, USA. *Soil and Tillage Research*. **165**, 325–335 (2017).

66.  Z. Luo, E. Wang, O. J. Sun, Can no-tillage stimulate carbon sequestration in agricultural soils? A meta-analysis of paired experiments. *Agriculture, Ecosystems & Environment*. **139**, 224–231 (2010).

67.  S. M. Ogle, C. Alsaker, J. Baldock, M. Bernoux, F. J. Breidt, B. McConkey, K. Regina, G. G. Vazquez-Amabile, Climate and Soil Characteristics Determine Where No-Till Management Can Store Carbon in Soils and Mitigate Greenhouse Gas Emissions. *Sci Rep*. **9**, 11665 (2019).

68.  J. W. Wendt, S. Hauser, An equivalent soil mass procedure for monitoring soil organic carbon in multiple soil layers. *European Journal of Soil Science*. **64**, 58–65 (2013).

69.  IPCC, 2019 Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories (2019), (available at https://www.ipcc-nggip.iges.or.jp/public/2019rf/vol4.html).

70.  W. H. Schlesinger, R. Amundson, *Glob Change Biol*, in press, doi:10.1111/gcb.14478.

71.  C. Chenu, D. A. Angers, P. Barré, D. Derrien, D. Arrouays, J. Balesdent, Increasing organic stocks in agricultural soils: Knowledge gaps and potential innovations. *Soil and Tillage Research*. **188**, 41–52 (2019).

72.  J. Zelikova, F. Chay, J. Freeman, D. Cullenward, A buyer's guide to soil carbon offsets, (available at https://carbonplan.org/research/soil-protocols-explainer).

73.  C. A. Sierra, A. M. Hoyt, Y. He, S. E. Trumbore, Soil Organic Matter Persistence as a Stochastic Process: Age and Transit Time Distributions of Carbon in Soils. *Global Biogeochem. Cycles*. **32**, 1574–1588 (2018).

74.  M. W. I. Schmidt, M. S. Torn, S. Abiven, T. Dittmar, G. Guggenberger, I. A. Janssens, M. Kleber, I. Kögel-Knabner, J. Lehmann, D. A. C. Manning, P. Nannipieri, D. P. Rasse, S. Weiner, S. E.

Trumbore, Persistence of soil organic matter as an ecosystem property. *Nature*. **478**, 49–56 (2011).

75. J. L. Soong, C. Castanha, C. E. Hicks Pries, N. Ofiti, R. C. Porras, W. J. Riley, M. W. I. Schmidt, M. S. Torn, Five years of whole-soil warming led to loss of subsoil carbon stocks and increased $CO_2$ efflux. *Sci. Adv.* **7**, eabd1343 (2021).

76. S. E. Baker, J. K. Stolaroff, G. Peridas, S. H. Pang, H. M. Goldstein, F. R. Lucci, W. Li, E. W. Slessarev, J. Pett-Ridge, F. J. Ryerson, J. L. Wagoner, W. Kirkendall, R. D. Aines, D. L. Sanchez, B. Cabiyo, J. Baker, S. McCoy, S. Uden, R. Runnebaum, J. Wilcox, P. C. Psarras, H. Pilorge, N. McQueen, D. Maynard, C. McCormick, "Getting to Neutral: Options for Negative Carbon Emissions in California" (LLNL-TR-796100, 1597217, 2019), p. LLNL-TR-796100, 1597217, , doi:10.2172/1597217.

77. J. Jian, X. Du, M. S. Reiter, R. D. Stewart, A meta-analysis of global cropland soil carbon changes due to cover cropping. *Soil Biology and Biochemistry*. **143**, 107735 (2020).

78. N. E. Tautges, J. L. Chiartas, A. C. M. Gaudin, A. T. O'Geen, I. Herrera, K. M. Scow, Deep soil inventories reveal that impacts of cover crops and compost on soil carbon sequestration differ in surface and subsurface soils. *Global Change Biology* (2019), doi:10.1111/gcb.14762.

79. M. Abdalla, A. Hastings, K. Cheng, Q. Yue, D. Chadwick, M. Espenberg, J. Truu, R. M. Rees, P. Smith, A critical review of the impacts of cover crops on nitrogen leaching, net greenhouse gas balance and crop productivity. *Glob Change Biol*. **25**, 2530–2543 (2019).

80. B. Guenet, B. Gabrielle, C. Chenu, D. Arrouays, J. Balesdent, M. Bernoux, E. Bruni, J. Caliman, R. Cardinael, S. Chen, P. Ciais, D. Desbois, J. Fouche, S. Frank, C. Henault, E. Lugato, V. Naipal, T. Nesme, M. Obersteiner, S. Pellerin, D. S. Powlson, D. P. Rasse, F. Rees, J. Soussana, Y. Su, H. Tian, H. Valin, F. Zhou, Can $N_2O$ emissions offset the benefits from soil organic carbon storage? *Glob. Change Biol.* **27**, 237–256 (2021).

81. I. Muhammad, U. M. Sainju, F. Zhao, A. Khan, R. Ghimire, X. Fu, J. Wang, Regulation of soil CO2 and N2O emissions by cover crops: A meta-analysis. *Soil and Tillage Research*. **192**, 103–112 (2019).

82. A. D. Basche, F. E. Miguez, T. C. Kaspar, M. J. Castellano, Do cover crops increase or decrease nitrous oxide emissions? A meta-analysis. *Journal of Soil and Water Conservation*. **69**, 471–482 (2014).

83. E. Lugato, A. Leip, A. Jones, Mitigation potential of soil carbon management overestimated by neglecting N2O emissions. *Nature Clim Change*. **8**, 219–223 (2018).

84. J. S. Bergtold, S. Ramsey, L. Maddy, J. R. Williams, A review of economic considerations for cover crops as a conservation practice. *Renew. Agric. Food Syst.* **34**, 62–76 (2019).

85. C. Tonitto, M. B. David, L. E. Drinkwater, Replacing bare fallows with cover crops in fertilizer-intensive cropping systems: A meta-analysis of crop yield and N dynamics. *Agriculture, Ecosystems & Environment*. **112**, 58–72 (2006).

86. C. A. Seifert, G. Azzari, D. B. Lobell, Satellite detection of cover crops and their effects on crop yield in the Midwestern United States. *Environ. Res. Lett.* **13**, 064033 (2018).

87. B. C. Runck, C. K. Khoury, P. M. Ewing, M. Kantar, The hidden land use cost of upscaling cover crops. *Commun Biol*. **3**, 300 (2020).

88. A. Plastina, F. Liu, F. Miguez, S. Carlson, Cover crops use in Midwestern US agriculture: perceived benefits and net returns. *Renew. Agric. Food Syst.* **35**, 38–48 (2020).

89. M. F. Cotrufo, J. L. Soong, A. J. Horton, E. E. Campbell, M. L. Haddix, D. H. Wall, W. J. Parton, Formation of soil organic matter via biochemical and physical pathways of litter mass loss. *Nature Geosci*. **8**, 776–779 (2015).

90. A. Ledo, P. Smith, A. Zerihun, J. Whitaker, J. L. Vicente-Vicente, Z. Qin, N. P. McNamara, Y. L. Zinn, M. Llorente, M. Liebig, M. Kuhnert, M. Dondini, A. Don, E. Diaz-Pines, A. Datta, H. Bakka, E. Aguilera, J. Hillier, *Glob Change Biol*, in press, doi:10.1111/gcb.15120.

91. R. T. Conant, C. E. P. Cerri, B. B. Osborne, K. Paustian, Grassland management impacts on soil carbon stocks: a new synthesis. *Ecological Applications*. **27**, 662–668 (2017).

92. I. Kämpf, N. Hölzel, M. Störrle, G. Broll, K. Kiehl, Potential of temperate agricultural soils for carbon sequestration: A meta-analysis of land-use effects. *Science of The Total Environment*. **566–567**, 428–435 (2016).

93. C. Li, L. M. Fultz, J. Moore-Kucera, V. Acosta-Martínez, J. Horita, R. Strauss, J. Zak, F. Calderón, D. Weindorf, Soil carbon sequestration potential in semi-arid grasslands in the Conservation Reserve Program. *Geoderma*. **294**, 80–90 (2017).

94. M. A. Liebig, M. R. Schmer, K. P. Vogel, R. B. Mitchell, Soil Carbon Storage by Switchgrass Grown for Bioenergy. *Bioenerg. Res.* **1**, 215–222 (2008).

95. E. W. Slessarev, E. E. Nuccio, K. J. McFarlane, C. Ramon, M. Saha, M. K. Firestone, J. Pett-Ridge, *GCB Bioenergy*, in press, doi:10.1111/gcbb.12729.

96. Y. Yang, T. Loecke, J. Knops, "Surface Soil Organic Carbon Sequestration Under Post Agricultural Grasslands Offset by Net Loss at Depth" (preprint, In Review, 2021), , doi:10.21203/rs.3.rs-449031/v1.

97. S. L. O'Brien, J. D. Jastrow, Physical and chemical protection in hierarchical soil aggregates regulates soil carbon and nitrogen recovery in restored perennial grasslands. *Soil Biology and Biochemistry*. **61**, 1–13 (2013).

98. D. L. Mummey, J. L. Smith, G. Bluhm, Assessment of alternative soil management practices on N2O emissions from US agriculture. *Agriculture, Ecosystems & Environment*. **70**, 79–87 (1998).

99. J. M. F. Johnson, N. W. Barbour, Nitrous Oxide Emission and Soil Carbon Sequestration from Herbaceous Perennial Biofuel Feedstocks. *Soil Science Society of America Journal*. **80**, 1057–1070 (2016).

100. R. J. Plevin, A. D. Jones, M. S. Torn, H. K. Gibbs, Greenhouse Gas Emissions from Biofuels' Indirect Land Use Change Are Uncertain but May Be Much Greater than Previously Estimated. *Environ. Sci. Technol.* **44**, 8015–8021 (2010).

101. I. Gelfand, R. Sahajpal, X. Zhang, R. C. Izaurralde, K. L. Gross, G. P. Robertson, Sustainable bioenergy production from marginal lands in the US Midwest. *Nature*. **493**, 514–517 (2013).

102. M. Abraha, I. Gelfand, S. K. Hamilton, J. Chen, G. P. Robertson, Carbon debt of field-scale conservation reserve program grasslands converted to annual and perennial bioenergy crops. *Environ. Res. Lett.* **14**, 024019 (2019).

103. S. Trumbore, Radiocarbon and Soil Carbon Dynamics. *Annu. Rev. Earth Planet. Sci.* **37**, 47–66 (2009).

104. J. A. Mathieu, C. Hatté, J. Balesdent, É. Parent, Deep soil carbon dynamics are driven more by soil type than by climate: a worldwide meta-analysis of radiocarbon profiles. *Glob Change Biol*. **21**, 4278–4292 (2015).

105. W. T. Baisden, R. L. Parfitt, Bomb 14C enrichment indicates decadal C pool in deep soil? *Biogeochemistry*. **85**, 59–68 (2007).

106. S. Fontaine, S. Barot, P. Barré, N. Bdioui, B. Mary, C. Rumpel, Stability of organic carbon in deep soil layers controlled by fresh carbon supply. *Nature*. **450**, 277–280 (2007).

107. W. C. Moldenhauer, G. W. Langdale, W. Frye, D. K. McCool, R. I. Papendick, D. E. Smika, W. Fryrear, Conservation tillage for erosion control. *Journal of Soil and Water Conservation*. **38**, 144–151 (1983).

108. L. Xiao, S. Zhou, R. Zhao, P. Greenwood, N. J. Kuhn, Evaluating soil organic carbon stock changes induced by no-tillage based on fixed depth and equivalent soil mass approaches. *Agriculture, Ecosystems & Environment*. **300**, 106982 (2020).

109. N. R. Haddaway, K. Hedlund, L. E. Jackson, T. Kätterer, E. Lugato, I. K. Thomsen, H. B. Jørgensen, P.-E. Isberg, How does tillage intensity affect soil organic carbon? A systematic review. *Environ Evid*. **6**, 30 (2017).

110. W. Sun, J. G. Canadell, L. Yu, L. Yu, W. Zhang, P. Smith, T. Fischer, Y. Huang, Climate drives global soil carbon sequestration and crop yield changes under conservation agriculture. *Glob Change Biol*. **26**, 3325–3335 (2020).

111. S. Mondal, D. Chakraborty, K. Bandyopadhyay, P. Aggarwal, D. S. Rana, A global analysis of the impact of zero-tillage on soil physical condition, organic carbon content, and plant root response. *Land Degrad Dev*. **31**, 557–567 (2020).

112. Z. Du, D. A. Angers, T. Ren, Q. Zhang, G. Li, The effect of no-till on organic C storage in Chinese soils should not be overemphasized: A meta-analysis. *Agriculture, Ecosystems & Environment*. **236**, 1–11 (2017).

113. K. H. E. Meurer, N. R. Haddaway, M. A. Bolinder, T. Kätterer, Tillage intensity affects total SOC stocks in boreo-temperate regions only in the topsoil—A systematic review using an ESM approach. *Earth-Science Reviews*. **177**, 613–622 (2018).

114. Y. Li, Z. Li, S. X. Chang, S. Cui, S. Jagadamma, Q. Zhang, Y. Cai, Residue retention promotes soil carbon accumulation in minimum tillage systems: Implications for conservation agriculture. *Science of The Total Environment*. **740**, 140147 (2020).

115. M. R. Nunes, D. L. Karlen, K. S. Veum, T. B. Moorman, C. A. Cambardella, Biological soil health indicators respond to tillage intensity: A US meta-analysis. *Geoderma*. **369**, 114335 (2020).

116. R. S. Nicoloso, C. W. Rice, *Soil Sci. Soc. Am. j.*, in press, doi:10.1002/saj2.20260.

117. X. Bai, Y. Huang, W. Ren, M. Coyne, P. Jacinthe, B. Tao, D. Hui, J. Yang, C. Matocha, Responses of soil carbon sequestration to climate-smart agriculture practices: A meta-analysis. *Glob Change Biol*. **25**, 2591–2606 (2019).

118. Y. Huang, W. Ren, L. Wang, D. Hui, J. H. Grove, X. Yang, B. Tao, B. Goff, Greenhouse gas emissions and crop yield in no-tillage systems: A meta-analysis. *Agriculture, Ecosystems & Environment*. **268**, 144–153 (2018).

119. J. Feng, F. Li, X. Zhou, C. Xu, L. Ji, Z. Chen, F. Fang, Impact of agronomy practices on the effects of reduced tillage systems on CH4 and N2O emissions from agricultural fields: A global meta-analysis. *PLoS ONE*. **13**, e0196703 (2018).

120. I. Virto, P. Barré, A. Burlot, C. Chenu, Carbon input differences as the main factor explaining the variability in soil organic C storage in no-tilled compared to inversion tilled agrosystems. *Biogeochemistry*. **108**, 17–26 (2012).

121. Roger Claasen, M. Bowman, J. McFadden, D. Smith, S. Wallander, Tillage Intensity and Conservation Cropping in the United States (2018).

122. H. Blanco-Canqui, C. S. Wortmann, Does occasional tillage undo the ecosystem services gained with no-till? A review. *Soil and Tillage Research*. **198**, 104534 (2020).

123. J. M. Baker, T. E. Ochsner, R. T. Venterea, T. J. Griffis, Tillage and soil carbon sequestration—What do we really know? *Agriculture, Ecosystems & Environment*. **118**, 1–5 (2007).

124. P. Poulton, J. Johnston, A. Macdonald, R. White, D. Powlson, Major limitations to achieving "4 per 1000" increases in soil organic carbon stock in temperate regions: Evidence from long-term experiments at Rothamsted Research, United Kingdom. *Glob Change Biol*. **24**, 2563–2584 (2018).

125. A. Charles, P. Rochette, J. K. Whalen, D. A. Angers, M. H. Chantigny, N. Bertrand, Global nitrous oxide emission factors from agricultural soils after addition of organic amendments: A meta-analysis. *Agriculture, Ecosystems & Environment*. **236**, 88–98 (2017).

126. M. S. DeLonge, R. Ryals, W. L. Silver, A Lifecycle Model to Evaluate Carbon Sequestration Potential and Greenhouse Gas Dynamics of Managed Grasslands. *Ecosystems*. **16**, 962–979 (2013).

127. D. S. Powlson, A. P. Whitmore, K. W. T. Goulding, Soil carbon sequestration to mitigate climate change: a critical re-examination to identify the true and the false. *European Journal of Soil Science*. **62**, 42–55 (2011).

128. J. Falcone, "Open-File Report," *Open-File Report* (Open-File Report, 2021).

129. M. H. Gerzabek, G. Haberhauer, H. Kirchmann, Soil Organic Matter Pools and Carbon-13 Natural Abundances in Particle-Size Fractions of a Long-Term Agricultural Field Experiment Receiving Organic Amendments. *Soil Sci. Soc. Am. J.* **65**, 352–358 (2001).

130. M. H. Gerzabek, F. Pichlmayer, H. Kirchmann, G. Haberhauer, The response of soil organic matter to manure amendments in a long-term experiment at Ultuna, Sweden. *European Journal of Soil Science*. **48**, 273–282 (1997).

131. R. Ryals, W. L. Silver, Effects of organic matter amendments on net primary productivity and greenhouse gas emissions in annual grasslands. *Ecological Applications*. **23**, 46–59 (2013).

132. L. Lai, S. Kumar, A global meta-analysis of livestock grazing impacts on soil properties. *PLoS ONE*. **15**, e0236638 (2020).

133. S. Mosier, S. Apfelbaum, P. Byck, F. Calderon, R. Teague, R. Thompson, M. F. Cotrufo, Adaptive multi-paddock grazing enhances soil carbon and nitrogen stocks and stabilization through mineral association in southeastern U.S. grazing lands. *Journal of Environmental Management*. **288**, 112409 (2021).

134. J. S. Sanderson, C. Beutler, J. R. Brown, I. Burke, T. Chapman, R. T. Conant, J. D. Derner, M. Easter, S. D. Fuhlendorf, G. Grissom, J. E. Herrick, D. Liptzin, J. A. Morgan, R. Murph, C. Pague, I. Rangwala, D. Ray, R. Rondeau, T. Schulz, T. Sullivan, Cattle, conservation, and carbon in the western Great Plains. *Journal of Soil and Water Conservation*. **75**, 5A-12A (2020).

135. D. E. Johnson, G. M. Ward, Estimates of animal methane emissions. *Environ Monit Assess*. **42**, 133–141 (1996).

136. O. Oenema, G. L. Velthof, S. Yamulki, S. C. Jarvis, Nitrous oxide emissions from grazed grassland. *Soil Use & Management*. **13**, 288–295 (1997).

137. L. G. Oates, R. D. Jackson, Livestock Management Strategy Affects Net Ecosystem Carbon Balance of Subhumid Pasture. *Rangeland Ecology & Management*. **67**, 19–29 (2014).

138. S. S. Paul, N. C. Coops, M. S. Johnson, M. Krzic, S. M. Smukler, Evaluating sampling efforts of standard laboratory analysis and mid-infrared spectroscopy for cost effective digital soil mapping at field scale. *Geoderma*. **356**, 113925 (2019).

139. R. A. Viscarra Rossel, D. J. Brus, The cost-efficiency and reliability of two methods for soil organic C accounting: The cost-efficiency and reliability of two methods for soil organic C accounting. *Land Degrad. Develop.* **29**, 506–520 (2018).

140. A. J. DeVincentis, S. S. Solis, E. M. Bruno, A. Leavitt, A. Gomes, S. Rice, D. Zaccaria, Using cost-benefit analysis to understand adoption of winter cover cropping in California's specialty crop systems. *Journal of Environmental Management*. **261**, 110205 (2020).

141. B. C. Murray, B. Sohngen, M. T. Ross, Economic consequences of consideration of permanence, leakage and additionality for soil carbon sequestration projects. *Climatic Change*. **80**, 127–143 (2007).

142. D. Cullenward, J. Hamman, J. Freeman, The cost of temporary carbon removal, (available at https://carbonplan.org/research/permanence-calculator-explainer).

143. J. Emmerling, L. Drouet, K.-I. van der Wijst, D. van Vuuren, V. Bosetti, M. Tavoni, The role of the discount rate for emission pathways and negative emissions. *Environ. Res. Lett.* **14**, 104008 (2019).

144. S. C. McClelland, K. Paustian, M. E. Schipanski, Management of cover crops in temperate climates influences soil organic carbon stocks: a meta-analysis. *Ecol. Appl.* **31** (2021), doi:10.1002/eap.2278.

145. T. Angelopoulou, N. Tziolas, A. Balafoutis, G. Zalidis, D. Bochtis, Remote Sensing Techniques for Soil Organic Carbon Estimation: A Review. *Remote Sensing*. **11**, 676 (2019).

146. J. Schimel, The implications of exoenzyme activity on microbial carbon and nitrogen limitation in soil: a theoretical model. *Soil Biology and Biochemistry*. **35**, 549–563 (2003).

147. W. R. Wieder, G. B. Bonan, S. D. Allison, Global soil carbon projections are improved by modelling microbial processes. *Nature Clim Change*. **3**, 909–912 (2013).

148. M. E. Craig, M. A. Mayes, B. N. Sulman, A. P. Walker, Biological mechanisms may contribute to soil carbon saturation patterns. *Glob Change Biol*. **27**, 2633–2644 (2021).

149. Y. He, S. E. Trumbore, M. S. Torn, J. W. Harden, L. J. S. Vaughn, S. D. Allison, J. T. Randerson, Radiocarbon constraints imply reduced carbon uptake by soils during the 21st century. *Science*. **353**, 1419–1424 (2016).

150. S. C. McClelland, K. Paustian, S. Williams, M. E. Schipanski, Modeling cover crop biomass production and related emissions to improve farm-scale decision-support tools. *Agricultural Systems*. **191**, 103151 (2021).

151. B. N. Sulman, J. A. M. Moore, R. Abramoff, C. Averill, S. Kivlin, K. Georgiou, B. Sridhar, M. D. Hartman, G. Wang, W. R. Wieder, M. A. Bradford, Y. Luo, M. A. Mayes, E. Morrison, W. J. Riley, A. Salazar, J. P. Schimel, J. Tang, A. T. Classen, Multiple models and experiments underscore large uncertainty in soil carbon dynamics. *Biogeochemistry*. **141**, 109–123 (2018).

152. D. R. Cameron, D. C. Marvin, J. M. Remucal, M. C. Passero, Ecosystem management and land conservation can substantially contribute to California's climate mitigation goals. *Proc Natl Acad Sci USA*. **114**, 12833–12838 (2017).

153.  N. L. Stephenson, A. J. Das, R. Condit, S. E. Russo, P. J. Baker, N. G. Beckman, D. A. Coomes, E. R. Lines, W. K. Morris, N. Rüger, E. Álvarez, C. Blundo, S. Bunyavejchewin, G. Chuyong, S. J. Davies, Á. Duque, C. N. Ewango, O. Flores, J. F. Franklin, H. R. Grau, Z. Hao, M. E. Harmon, S. P. Hubbell, D. Kenfack, Y. Lin, J.-R. Makana, A. Malizia, L. R. Malizia, R. J. Pabst, N. Pongpattananurak, S.-H. Su, I.-F. Sun, S. Tan, D. Thomas, P. J. van Mantgem, X. Wang, S. K. Wiser, M. A. Zavala, Rate of tree carbon accumulation increases continuously with tree size. *Nature*. **507**, 90–93 (2014).

154.  G. M. Domke, S. N. Oswalt, B. F. Walters, R. S. Morin, Tree planting has the potential to increase carbon sequestration capacity of forests in the United States. *Proc Natl Acad Sci USA*. **117**, 24649–24651 (2020).

155.  J. E. Fargione, S. Bassett, T. Boucher, S. D. Bridgham, R. T. Conant, S. C. Cook-Patton, P. W. Ellis, A. Falcucci, J. W. Fourqurean, T. Gopalakrishna, H. Gu, B. Henderson, M. D. Hurteau, K. D. Kroeger, T. Kroeger, T. J. Lark, S. M. Leavitt, G. Lomax, R. I. McDonald, J. P. Megonigal, D. A. Miteva, C. J. Richardson, J. Sanderman, D. Shoch, S. A. Spawn, J. W. Veldman, C. A. Williams, P. B. Woodbury, C. Zganjar, M. Baranski, P. Elias, R. A. Houghton, E. Landis, E. McGlynn, W. H. Schlesinger, J. V. Siikamaki, A. E. Sutton-Grier, B. W. Griscom, Natural climate solutions for the United States. *Sci. Adv.* **4**, eaat1869 (2018).

156.  N. R. S. U.S. Forest Service, "Who Owns America's Forests? Forest Ownership Patterns and Family Forest Highlights from the National Woodland Owner Survey" (NRS-INF-06-08, 2008).

157.  W. A. Jenkins, L. P. Olander, B. C. Murray, Addressing leakage in a greenhouse gas mitigation offsets program for forestry and agriculture. *NI PB*, 09–03 (2009).

158.  B. Haya, Policy brief: The California Air Resources Board's US Forest offset protocol underestimates leakage. *Berkeley, CA* (2019).

159.  B. C. Murray, B. A. McCarl, H.-C. Lee, Estimating Leakage from Forest Carbon Sequestration Programs. *Land Economics*. **80**, 109–124 (2004).

160.  J. Gan, B. A. McCarl, Measuring transnational leakage of forest conservation. *Ecological Economics*. **64**, 423–432 (2007).

161.  B. Sun, B. Sohngen, Set-asides for carbon sequestration: implications for permanence and leakage. *Climatic Change*. **96**, 409–419 (2009).

162.  A. Roopsind, B. Sohngen, J. Brandt, Evidence that a national REDD+ program reduces tree cover loss and carbon emissions in a high forest cover, low deforestation country. *Proc Natl Acad Sci USA*. **116**, 24492–24499 (2019).

163.  D. Haim, E. M. White, R. J. Alig, Agriculture Afforestation for Carbon Sequestration Under Carbon Markets in the United States: Leakage Behavior from Regional Allowance Programs. *Applied Economic Perspectives and Policy*. **38**, 132–151 (2016).

164.  G. Badgley, J. Freeman, J. J. Hamman, B. Haya, A. T. Trugman, W. R. Anderegg, D. Cullenward, Systematic over-crediting in California's forest carbon offsets program. *bioRxiv* (2021).

165. E. S. Sanz-Pérez, C. R. Murdock, S. A. Didas, C. W. Jones, Direct Capture of CO2 from Ambient Air. *Chem. Rev.* **116**, 11840–11876 (2016).

166. S. Choi, J. H. Drese, P. M. Eisenberger, C. W. Jones, Application of Amine-Tethered Solid Sorbents for Direct CO2 Capture from the Ambient Air. *Environ. Sci. Technol.* **45**, 2420–2427 (2011).

167. A. Goeppert, M. Czaun, R. B. May, G. K. S. Prakash, G. A. Olah, S. R. Narayanan, Carbon Dioxide Capture from the Air Using a Polyamine Based Regenerable Solid Adsorbent. *J. Am. Chem. Soc.* **133**, 20164–20167 (2011).

168. S. A. Didas, S. Choi, W. Chaikittisilp, C. W. Jones, Amine–Oxide Hybrid Materials for CO2 Capture from Ambient Air. *Acc. Chem. Res.* **48**, 2680–2687 (2015).

169. M. A. Sakwa-Novak, C.-J. Yoo, S. Tan, F. Rashidi, C. W. Jones, Poly(ethylenimine)-Functionalized Monolithic Alumina Honeycomb Adsorbents for CO2 Capture from Air. *ChemSusChem*. **9**, 1859–1868 (2016).

170. Climeworks, (available at https://climeworks.com).

171. Global Thermostat, (available at https://globalthermostat.com/).

172. S. Choi, T. Watanabe, T.-H. Bae, D. S. Sholl, C. W. Jones, Modification of the Mg/DOBDC MOF with Amines to Enhance CO2 Adsorption from Ultradilute Gases. *J. Phys. Chem. Lett.* **3**, 1136–1141 (2012).

173. T. M. McDonald, W. R. Lee, J. A. Mason, B. M. Wiers, C. S. Hong, J. R. Long, Capture of Carbon Dioxide from Air and Flue Gas in the Alkylamine-Appended Metal–Organic Framework mmen-Mg2(dobpdc). *J. Am. Chem. Soc.* **134**, 7056–7065 (2012).

174. O. Shekhah, Y. Belmabkhout, Z. Chen, V. Guillerm, A. Cairns, K. Adil, M. Eddaoudi, Made-to-order metal-organic frameworks for trace carbon dioxide removal and air capture. *Nat Commun*. **5**, 4228 (2014).

175. M. M. Sadiq, M. P. Batten, X. Mulet, C. Freeman, K. Konstas, J. I. Mardel, J. Tanner, D. Ng, X. Wang, S. Howard, M. R. Hill, A. W. Thornton, A Pilot-Scale Demonstration of Mobile Direct Air Capture Using Metal-Organic Frameworks. *Advanced Sustainable Systems*. **n/a**, 2000101 (2020).

176. N. McQueen, P. Kelemen, G. Dipple, P. Renforth, J. Wilcox, Ambient weathering of magnesium oxide for CO2 removal from air. *Nature Communications*. **11**, 3299 (2020).

177. Heirloom, (available at https://www.heirloomcarbon.com/).

178. T. Wang, K. S. Lackner, A. Wright, Moisture Swing Sorbent for Carbon Dioxide Capture from Ambient Air. *Environ. Sci. Technol.* **45**, 6670–6675 (2011).

179. X. Shi, H. Xiao, K. Kanamori, A. Yonezu, K. S. Lackner, X. Chen, Moisture-Driven CO2 Sorbents. *Joule* (2020), doi:10.1016/j.joule.2020.07.005.

180. Carbon Collect, (available at https://mechanicaltrees.com/).

181. S. Voskian, T. A. Hatton, Faradaic electro-swing reactive adsorption for CO2 capture. *Energy Environ. Sci.* (2019), doi:10.1039/C9EE02412C.

182. Verdox, (available at https://www.verdox.com).

183. D. W. Keith, G. Holmes, D. St. Angelo, K. Heidel, A Process for Capturing CO2 from the Atmosphere. *Joule*. **2**, 1573–1594 (2018).

184. F. M. Brethomé, N. J. Williams, C. A. Seipp, M. K. Kidder, R. Custelcean, Direct air capture of CO 2 via aqueous-phase absorption and crystalline-phase release using concentrated solar power. *Nature Energy*. **3**, 553 (2018).

185. R. Custelcean, K. A. Garrabrant, P. Agullo, N. J. Williams, Direct air capture of CO2 with aqueous peptides and crystalline guanidines. *Cell Reports Physical Science*. **2**, 100385 (2021).

186. M. D. Eisaman, L. Alvarado, D. Larner, P. Wang, B. Garg, K. A. Littau, CO 2 separation using bipolar membrane electrodialysis. *Energy & Environmental Science*. **4**, 1319–1328 (2011).

187. F. Sabatino, M. Mehta, A. Grimm, M. Gazzani, F. Gallucci, G. J. Kramer, M. van Sint Annaland, Evaluation of a Direct Air Capture Process Combining Wet Scrubbing and Bipolar Membrane Electrodialysis. *Ind. Eng. Chem. Res.* **59**, 7007–7020 (2020).

188. I. A. Digdaya, I. Sullivan, M. Lin, L. Han, W.-H. Cheng, H. A. Atwater, C. Xiang, A direct coupled electrochemical system for capture and conversion of CO 2 from oceanwater. *Nature Communications*. **11**, 4412 (2020).

189. S. Jin, M. Wu, R. G. Gordon, M. J. Aziz, D. G. Kwabi, pH swing cycle for CO 2 capture electrochemically driven through proton-coupled electron transfer. *Energy & Environmental Science*. **13**, 3706–3722 (2020).

190. M. Rahimi, G. Catalini, M. Puccini, T. A. Hatton, Bench-scale demonstration of CO2 capture with an electrochemically driven proton concentration process. *RSC Adv.* **10**, 16832–16843 (2020).

191. Q. Shu, L. Legrand, P. Kuntke, M. Tedesco, H. V. M. Hamelers, Electrochemical Regeneration of Spent Alkaline Absorbent from Direct Air Capture. *Environ. Sci. Technol.* (2020), doi:10.1021/acs.est.0c01977.

192. N. McQueen, P. Psarras, H. Pilorgé, S. Liguori, J. He, M. Yuan, C. M. Woodall, K. Kian, L. Pierpoint, J. Jurewicz, J. M. Lucas, R. Jacobson, N. Deich, J. Wilcox, Cost Analysis of Direct Air Capture and Sequestration Coupled to Low-Carbon Thermal Energy in the United States. *Environ. Sci. Technol.* (2020), doi:10.1021/acs.est.0c00476.

193. APS Physics, "Direct Air Capture of CO2 with Chemicals" (American Physical Society, 2011).

194. National Academies of Sciences, Engineering, and Medicine, *Negative Emissions Technologies and Reliable Sequestration: A Research Agenda* (2018; https://www.nap.edu/catalog/25259/negative-emissions-technologies-and-reliable-sequestration-a-research-agenda).

195. H. Azarabadi, K. S. Lackner, A sorbent-focused techno-economic analysis of direct air capture. *Applied Energy*. **250**, 959–975 (2019).

196. D. DeSantis, J. A. Mason, B. D. James, C. Houchins, J. R. Long, M. Veenstra, Techno-economic Analysis of Metal–Organic Frameworks for Hydrogen and Natural Gas Storage. *Energy Fuels*. **31**, 2024–2032 (2017).

197. Calix, (available at https://www.calix.global/our-technology/).

198. B. Pivovar, Y. S. Kim, "2019 Anion Exchange Membrane Workshop Summary Report" (NREL/TP-5900-77240, National Renewable Energy Lab. (NREL), Golden, CO (United States), 2020), , doi:10.2172/1660106.

199. E. S. Rubin, I. M. L. Azevedo, P. Jaramillo, S. Yeh, A review of learning rates for electricity supply technologies. *Energy Policy*. **86**, 198–218 (2015).

200. S. E. Baker, J. K. Stolaroff, G. Peridas, S. H. Pang, H. M. Goldstein, F. R. Lucci, W. Li, E. W. Slessarev, J. Pett-Ridge, F. J. Ryerson, J. L. Wagoner, W. Kirkendall, R. D. Aines, D. L. Sanchez, B. Cabiyo, J. Baker, S. T. McCoy, S. Uden, R. Runnebaum, J. Wilcox, P. C. Psarras, H. Pilorge, N. McQueen, D. Maynard, C. McCormick, "Getting to Neutral: Options for Negative Carbon Emissions in California" (LLNL-TR-796100, Lawrence Livermore National Laboratory, 2020), (available at https://www-gs.llnl.gov/content/assets/docs/energy/Getting_to_Neutral.pdf).

201. J. Larsen, W. Herndon, M. Grant, P. Marsters, "Capturing Leadership: Policies for the US to Advance Direct Air Capture Technology" (Rhodium Group, LLC, 2019), (available at https://rhg.com/research/capturing-leadership-policies-for-the-us-to-advance-direct-air-capture-technology/).

202. S. A. Wilson, Verifying and quantifying carbon fixation in minerals from serpentine-rich mine tailings using the Rietveld method with X-ray powder diffraction data. *American Mineralogist*. **91**, 1331–1341 (2006).

203. P. Ciais, C. Sabine, G. Bala, L. Bopp, V. Brovkin, J. Canadell, A. Chhabra, R. DeFries, J. Galloway, M. Heimann, C. Jones, C. Le Quéré, R. B. Myneni, S. Piao, P. Thornton, "Carbon and Other Biogeochemical Cycles. In: Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Stocker, T.F., D. Qin, G.-K. Plattner, M. Tignor, S.K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex and P.M. Midgley (eds.)]" (Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA., 2013).

204. P. Kelemen, S. M. Benson, H. Pilorgé, P. Psarras, J. Wilcox, An Overview of the Status and Challenges of CO2 Storage in Minerals and Geological Formations. *Frontiers in Climate*. **1**, 9 (2019).

205. B. P. McGrail, F. A. Spane, E. C. Sullivan, D. H. Bacon, G. Hund, The Wallula basalt sequestration pilot project. *Energy Procedia*. **4**, 5653–5660 (2011).

206. B. P. McGrail, H. T. Schaef, F. A. Spane, J. A. Horner, A. T. Owen, J. B. Cliff, O. Qafoku, C. J. Thompson, E. C. Sullivan, Wallula Basalt Pilot Demonstration Project: Post-injection Results and Conclusions. *Energy Procedia*. **114**, 5783–5790 (2017).

207. S. K. White, F. A. Spane, H. T. Schaef, Q. R. S. Miller, M. D. White, J. A. Horner, B. P. McGrail, Quantification of $CO_2$ Mineralization at the Wallula Basalt Pilot Project. *Environ. Sci. Technol.* **54**, 14609–14616 (2020).

208. Carbfix, Our story. *Carbfix* (2021), (available at https://www.carbfix.com/our-story).

209. J. M. Matter, M. Stute, S. Ó. Snæbjörnsdottir, E. H. Oelkers, S. R. Gislason, E. S. Aradottir, B. Sigfusson, I. Gunnarsson, H. Sigurdardottir, E. Gunnlaugsson, G. Axelsson, H. A. Alfredsson, D. Wolff-Boenisch, K. Mesfin, D. F. de la R. Taya, J. Hall, K. Dideriksen, W. S. Broecker, Rapid carbon mineralization for permanent disposal of anthropogenic carbon dioxide emissions. *Science*. **352**, 1312–1314 (2016).

210. S. Ó. Snæbjörnsdóttir, E. H. Oelkers, K. Mesfin, E. S. Aradóttir, K. Dideriksen, I. Gunnarsson, E. Gunnlaugsson, J. M. Matter, M. Stute, S. R. Gislason, The chemistry and saturation states of subsurface fluids during the in situ mineralisation of $CO_2$ and $H_2S$ at the CarbFix site in SW-Iceland. *International Journal of Greenhouse Gas Control*. **58**, 87–102 (2017).

211. S. Ó. Snæbjörnsdóttir, S. R. Gislason, I. M. Galeczka, E. H. Oelkers, Reaction path modelling of in-situ mineralisation of $CO_2$ at the CarbFix site at Hellisheidi, SW-Iceland. *Geochimica et Cosmochimica Acta*. **220**, 348–366 (2018).

212. E. M. Mervine, S. A. Wilson, I. M. Power, G. M. Dipple, C. C. Turvey, J. L. Hamilton, S. Vanderzee, M. Raudsepp, C. Southam, J. M. Matter, P. B. Kelemen, J. Stiefenhofer, Z. Miya, G. Southam, Potential for offsetting diamond mine carbon emissions through mineral carbonation of processed kimberlite: an assessment of De Beers mine sites in South Africa and Canada. *Miner Petrol*. **112**, 755–765 (2018).

213. I. M. Power, J. McCutcheon, A. L. Harrison, S. A. Wilson, G. M. Dipple, S. Kelly, C. Southam, G. Southam, Strategizing Carbon-Neutral Mines: A Case for Pilot Projects. *Minerals*. **4**, 399–436 (2014).

214. I. M. Power, G. M. Dipple, P. M. D. Bradshaw, A. L. Harrison, Prospects for $CO_2$ mineralization and enhanced weathering of ultramafic mine tailings from the Baptiste nickel deposit in British Columbia, Canada. *International Journal of Greenhouse Gas Control*. **94**, 102895 (2020).

215. S. A. Wilson, A. L. Harrison, G. M. Dipple, I. M. Power, S. L. L. Barker, K. Ulrich Mayer, S. J. Fallon, M. Raudsepp, G. Southam, Offsetting of $CO_2$ emissions by air capture in mine tailings at the Mount Keith Nickel Mine, Western Australia: Rates, controls and prospects for carbon neutral mining. *International Journal of Greenhouse Gas Control*. **25**, 121–140 (2014).

216. D. M. Franks, M. Stringer, L. A. Torres-Cruz, E. Baker, R. Valenta, K. Thygesen, A. Matthews, J. Howchin, S. Barrie, Tailings facility disclosures reveal stability risks. *Sci Rep*. **11**, 5353 (2021).

217. U.S. Geological Survey, "Mineral Resources Data System" (U.S. Geological Survey, Reston, Virginia, 2005), (available at https://mrdata.usgs.gov/mrds/).

218.   G. T. Mason, R. E. Arndt, "Mineral Resources Data System (MRDS)," *USGS Data Series 20* (1996), (available at https://doi.org/10.3133/ds20).

219.   McFaul, E.J., Mason, G.T., Ferguson, W.B., Lipin, B.R., "U.S. Geological Survey mineral databases; MRDS and MAS/MILS," *USGS Data Series 52* (2000), (available at https://doi.org/10.3133/ds52).

220.   Schweitzer, P.N., "Record quality tables for the Mineral Resources Data System," *U.S. Geological Survey data release* (2019), (available at https://doi.org/10.5066/P9DYLWMP).

221.   I. M. Power, A. L. Harrison, G. M. Dipple, S. A. Wilson, P. B. Kelemen, M. Hitch, G. Southam, Carbon Mineralization: From Natural Analogues to Engineered Systems. *Reviews in Mineralogy and Geochemistry.* **77**, 305–360 (2013).

222.   Palandri, James L., Kharaka, Yousif K., "A compilation of rate parameters of water-mineral interaction kinetics for application to geochemical modeling" (U.S. Geological Survey, Menlo Park, CA, 2004).

223.   P. B. Kelemen, N. McQueen, J. Wilcox, P. Renforth, G. Dipple, A. P. Vankeuren, Engineered carbon mineralization in ultramafic rocks for $CO_2$ removal from air: Review and new insights. *Chemical Geology.* **550**, 119628 (2020).

224.   H. C. Oskierski, B. Z. Dlugogorski, G. Jacobsen, Sequestration of atmospheric $CO_2$ in chrysotile mine tailings of the Woodsreef Asbestos Mine, Australia: Quantitative mineralogy, isotopic fingerprinting and carbonation rates. *Chemical Geology.* **358**, 156–169 (2013).

225.   A. L. Harrison, I. M. Power, G. M. Dipple, Accelerated Carbonation of Brucite in Mine Tailings for Carbon Sequestration. *Environ. Sci. Technol.* **47**, 126–134 (2013).

226.   G. P. Assima, F. Larachi, G. Beaudoin, J. Molson, $CO_2$ Sequestration in Chrysotile Mining Residues—Implication of Watering and Passivation under Environmental Conditions. *Ind. Eng. Chem. Res.* **51**, 8726–8734 (2012).

227.   I. M. Power, A. L. Harrison, G. M. Dipple, Accelerating Mineral Carbonation Using Carbonic Anhydrase. *Environ. Sci. Technol.* **50**, 2610–2618 (2016).

228.   M. S. Ncongwane, J. L. Broadhurst, J. Petersen, Assessment of the potential carbon footprint of engineered processes for the mineral carbonation of PGM tailings. *International Journal of Greenhouse Gas Control.* **77**, 70–81 (2018).

229.   U.S. Geological Survey, "Mineral Commodity Summaries 2021" (U.S. Geological Survey, 2021), p. 200.

230.   S. C. Krevor, C. R. Graves, B. S. Van Gosen, A. E. McCafferty, "Mapping the mineral resource base for mineral carbon-dioxide sequestration in the conterminous United States," *U.S. Geological Survey Digital Data Series 414* (Data Series, U.S. Geological Survey, 2009), p. 14.

231.   C. Woodall, N. McQueen, J. Wilcox, "Carbon mineralization using plagioclase feldspar-rich mine tailings: characterization and cation extraction" (SSRN Scholarly Paper ID 3820013, Social Science Research Network, Rochester, NY, 2021), , doi:10.2139/ssrn.3820013.

232. Mosier, Dan L., Singer, D.A., Moring, B.C., Galloway, J.P., "Podiform chromite deposits—database and grade and tonnage models," *US Geological Survey Scientific Investigations Report 5157* (2012), p. 45.

233. G. A. Kingston, R. A. Miller, F. V. Carrillo, "Availability of U.S. Chromium Resources," *Information circular 8465* (U.S. Department of Interior, Bureau of Mines), p. 23 p., illus.tables.

234. Boyle, E. H., Jr, Shields, D.J., Wagner, L.A., "Chromium availability in market economy countries and network flow model analysis of world chromium supply," *U.S. Bureau of Mines Information Circular 9337* (1993), p. 131.

235. Indian Bureau of Mines, "Monograph on Chromite" (Controller General, Indian Bureau of Mines, 2013).

236. Kim, Jonathan, "Magnesium Extraction from Asbestos Mine Tailings: A Report" (Prepared for the Vermont Geologic Survey Department of Environmental Protection, 1998), p. 43.

237. Aines, R.A., "Personal communication" (2021).

238. Levin, M., Asbestos Workers Fear for Jobs, Not Health. *Los Angeles Times* (1986).

239. F. Larachi, I. Daldoul, G. Beaudoin, Fixation of CO2 by chrysotile in low-pressure dry and moist carbonation: Ex-situ and in-situ characterizations. *Geochimica et Cosmochimica Acta*. **74**, 3051–3075 (2010).

240. S. A. Wilson, G. M. Dipple, I. M. Power, J. M. Thom, R. G. Anderson, M. Raudsepp, J. E. Gabites, G. Southam, Carbon Dioxide Fixation within Mine Wastes of Ultramafic-Hosted Ore Deposits: Examples from the Clinton Creek and Cassiar Chrysotile Deposits, Canada. *Economic Geology*. **104**, 95–112 (2009).

241. S. J. Gerdemann, W. K. O'Connor, D. C. Dahlin, L. R. Penner, H. Rush, Ex Situ Aqueous Mineral Carbonation. *Environ. Sci. Technol.* **41**, 2587–2593 (2007).

242. Derkics, D., "Report to congress: wastes from the extraction and beneficiation of metallic ores, phosphate rock, asbestos, overburden from uranium mining, and oil shale" (Final report PB-88-162631/XAB; EPA-530/SW-85/033, Environmental Protection Agency, Office of Solid Waste, Washington, DC (USA), 1985).

243. U.S. Geological Survey, "Asbestos statistics [through 2018; last modified May 28, 2019], in Kelly, T.D., and Matos, G.R., comps., Historical statistics for mineral and material commodities in the United States (2016 version)," *U.S. Geological Survey Data Series 140* (2018), (available at https://www.usgs.gov/centers/nmic/historical-statistics-mineral-and-material-commodities-united-states#asbestos).

244. Virta, Robert L., "Mineral Commodity Profiles, Asbestos" (U.S. Geological Survey, Reston, VA, 2005).

245. Van Gosen, Bradley S., Clinkenbeard, John P., "Reported historic asbestos mines, historic asbestos prospects, and other natural occurrences of asbestos in California" (U.S. Geological Survey, 2011).

246. Mumpton, F. A., The stability of brucite in the weathering zone of the New Idria serpentinite. *Clays and Clay Minerals*, 249–257 (1966).

247. Chidester, A. H., "Petrology, structure, and genesis of the asbestos-bearing ultramafic rocks of the Belvidere Mountain area in Vermont" (U.S. Geological Survey, 1978), p. 95.

248. K. Lechat, J.-M. Lemieux, J. Molson, G. Beaudoin, R. Hébert, Field evidence of CO2 sequestration by mineral carbonation in ultramafic milling wastes, Thetford Mines, Canada. *International Journal of Greenhouse Gas Control*. **47**, 110–121 (2016).

249. C. Myers, T. Nakagaki, Direct mineralization of atmospheric CO2 using natural rocks in Japan. *Environ. Res. Lett.* **15**, 124018 (2020).

250. Carbfix, Coda Terminal. *Carbfix* (2021), (available at https://www.carbfix.com/codaterminal).

251. National Energy Technology Laboratory, "FE/NETL CO2 Saline Storage Cost Model. Last Update: Sep 2017 (Version 3)" (U.S. Department of Energy, 2017), (available at https://www.netl.doe.gov/research/energy-analysis/search-publications/vuedetails?id=2403).

252. E. S. Rubin, J. E. Davison, H. J. Herzog, The cost of CO2 capture and storage. *International Journal of Greenhouse Gas Control*. **40**, 378–400 (2015).

253. Blondes, M.S, Merrill, M.D., Anderson, S.T., DeVera, C.A., "Carbon dioxide mineralization feasibility in the United States," *Scientific Investigations Report 2018–5079* (U.S. Geological Survey, 2019), p. 29.